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16. ABSTRACT

This report reviews the literature concerning the consequences of nutrient enrichment, the significance of phosphorus in eutrophication, critical phosphorus concentrations, the effects of eutrophication on human populations, and the extent of the eutrophication problem in the United States. The evidence contained in this report indicates that (a) excessive nutrient concentrations are associated with undesirable changes (eutrophication) in aquatic plants, depletion of dissolved oxygen, disappearance of cold water fish, and appearance of nuisance algal species; (b) excessive phosphorus is most frequently responsible for these undesirable changes in lakes; (c) lakes and reservoirs respond more severely to excessive phosphorus concentrations than do flowing waters, and do so at lower phosphorus concentrations; (d) phosphorus may at times be the limiting factor in estuaries but is not usually the limiting factor in coastal waters; (e) critical phosphorus levels which lead to eutrophication have not been clearly defined because of the variation in the response of surface waters to phosphorus caused by differences in residence times, mixing, sunlight penetration, etc., although some guidelines for phosphorus loadings and concentrations have established; (f) eutrophication has adversely affected human populations through increased water treatment costs, decreased property values, changes in the commercial fishery and reduction of the aesthetic and recreational values of affected lakes.

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THE IMPACT OF INORGANIC PHOSPHATES IN THE ENVIRONMENT

FINAL REPORT

NOVEMBER, 1978

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NOTICE

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Conclusions

The evidence contained in this report indicates that (a) excessive nutrient concentrations are associated with undesirable changes (eutrophication) in aquatic ecosystems, including excessive production of aquatic plants, depletion of dissolved oxygen, disappearance of cold water fish, and appearance of nuisance algal species; (b) excessive phosphorus is most frequently responsible for these undesirable changes in lakes; (c) lakes and reservoirs respond more severely to excessive phosphorus concentrations than do flowing waters, and do so at lower phosphorus concentrations; (d) phosphorus may at times be the limiting factor in estuaries but is not usually the limiting factor in coastal waters; (e) critical phosphorus levels which lead to eutrophication have not been clearly defined because of the variation in the response of surface waters to phosphorus caused by differences in residence times, mixing, sunlight penetration, etc., although some guidelines for phosphorus loadings and concentrations have established; (f) eutrophication has adversely affected human populations through increased water treatment costs, decreased property values, changes in the commercial fishery and reduction of the aesthetic and recreational values of affected lakes.

Eutrophication caused by excessive phosphorus levels is a problem which occurs in most of the states, although the sources of excessive phosphorus vary substantially even within states, and has the potential to become more widespread. While phosphorus-caused eutrophication has not yet impacted all of the nation's surface waters, there is a clear potential for expansion of the problem without proper remedial and protective actions.

I. Introduction

Phosphorus problems in the environment are related to the fact that phosphorus is an essential plant nutrient. Under certain conditions, phosphorus enrichment can induce radical changes in aquatic ecosystems that impact human populations using the affected resource. This report characterizes the environmental problem associated with phosphate enrichment of surface waters. The problem can be defined as eutrophication, a term which describes the enrichment of surface waters with nutrients (primarily nitrogen and phosphorus) and the accompanying effects. It occurs naturally, with nutrients derived from decaying organic matter or phosphate deposits, or artifically, the product of excessive nutrient release from agricultural, industrial, or urban activities. Whatever the source, nutrient loadings are, in part, responsible for the productivity of surface waters. Phosphorus is, most often, the nutrient responsible for the eutrophication of freshwater lakes, and usually the nutrient which is easiest to control.

II. Consequences of Nutrient Enrichment

Effects at Specific Sites

Nutrients, along with other factors such as temperature and light, control the productivity of water bodies and to a large extent, the species composition and structure of aquatic communities. Special examination of several in-lake studies will illustrate some of the changes which occur during the eutrophication of lakes.

Schindler et al. (1971 and 1973) studied the effects of enrichment with known quantities of nutrients on a small, unproductive lake (lake 227) previously undisturbed by human activity. They added phosphorous (as $NaHPO_4$ or H_3PO_4) and nitrogen at levels which increased the loading by a factor of five. Fertilization took place for 17-21 weeks during the ice-free periods of 1969 through 1972. Phytoplankton standing crops, as measured by chlorophyll a content, increased in response to fertilization. At mid-summer, 1968, prior to fertilization, chlorophyll a concentrations did not exceed 3 ug/l. After fertilization, epilimnetic chlorophyll a levels in August ranged from 9-24 ug/l in 1969, 48-92 ug/l in 1970, 6-40 ug/l in 1971, 30-70 ug/l in 1972 (Schindler and Fee, 1974). Chlorophyll a maxima were dramatically higher after fertilization. Phytoplankton species also changed as a result of nitrogen and phosphorus fertilization. In 1968, the year prior to fertilization, pigmented flagellated algae (cryptophytes and chrysophytes) were dominant throughout the summer. After fertilization, chlorophytes, especially Scenedesmus, became the dominant group (Schindler, 1977), and blue-green algae (cyanophytes) became more abundant than before fertilization. No floating algal blooms were present during the first summer of fertilization, but during the second summer (Schindler, 1971), cyanophytes appeared in large numbers for the first time.

Langford (1948) and Smith (1969) studied the response to fertilization of lakes in another part of the Canadian Precambrian Shield. In both of these studies, the quantitative growth response of algae to nutrient inputs was similar to that found by Schindler et al. (1971). Algal species responses differed, however. While Schindler et al. saw an increase in Chlorophyta, Langford (1948) found diatoms of the genera Tabellaria, Fragilaria, and Melosira most responsive and Smith (1969) found increases in populations of the blue-green algae, Anabaena and Anacystis, and the green alga Spirogyra. The ratio of the various nutrient inputs, particularly N:P, is probably responsible for the varying response of phytoplankton groups (Schindler, 1977).

Vanderhoef et al. (1974) studied the phytoplankton nutrient concentrations at fourteen sites in Green Bay, Lake Michigan. The open end of the bay was dominated by diatoms (Fragilaria, Melosira, Tabellaria, Asterionella), but, as nutrient levels increased in the inner portions of the bay, blue-green algae predominated. Microcystis was favored in areas where both nitrogen and phosphorus were available. However, with the depletion of nitrogen relative to phosphorus levels, Aphanizomenon, a nitrogen-fixing alga, appeared to out-compete Microcystis.

A similar study was conducted in Provo Bay, Utah Lake, by Bradshaw et al. (1973). Nutrient concentrations decreased from 1.2 to 0.1 mg/l phosphorus and 0.7 to 0.1 mg/l nitrogen with movement out of the bay. Biochemical oxygen demand (BOD), algal standing crop (cells/liter) and chironomid larvae numbers were much higher in the bay than outside the bay where dilution of nutrients occurred. Turbidity caused by the presence of algal cells was higher in the bay than in the lake.

Lake Washington, Seattle, is a well-documented case illustrating the consequences of eutrophication, as well as an example of the effects of the nutrients in sewage effluents, and lake recovery

following sewage diversion (Edmondson 1968, 1970, and 1977). During the period between 1941 and 1963, treated sewage effluent entered the lake in increasing amounts. In 1957, the total phosphorus loading was between 100,000 and 108,000 kg, 42 percent (42,000 kg) of which was derived from sewage effluent. By 1959, 11 secondary treatment plants were operating, their effluent amounting to more than 23,200 m^3 per day. In 1962, total phosphorus inputs to the lake amounted to 231,000 kg of which 65 percent (150,000 kg) was contained in sewage effluent. By the early 1950s, Lake Washington was beginning to show signs of eutrophication, relative to conditions in 1933. Winter phosphorus concentrations had nearly doubled. The appearance of Oscillatoria rubescens, a colonial blue-green alga, in 1955, and oxygen depletion of the hypolimnion warned water quality managers of the continued deterioration of the lake. In 1958, a program to divert the sewage effluent to Puget Sound was funded by public vote. By 1963, one third of the sewage had been diverted, and effluents at this time totaled about 75,600 m^3 per day. Diversion was 99 percent complete by 1967. Figures 1 (Odum, 1971) and 2 (Edmondson, 1977) illustrate the important change in physical, chemical, and biological indicators. Transparency and diatom species diversity declined rapidly during the 1950s, while phosphate phosphorus and the percent of eutrophic species in the diatom community increased. During and after sewage diversion, these indices reversed their direction in response to decreased nutrient income (Odum, 1971). In this case, secchi disk transparency was a measure of algal standing crop rather than suspended silt. Before diversion the mean summer transparency decreased from 2.1 m in 1950, was lowest in 1963 (1 m), but had increased to 2.8 m by 1969 (Edmondson, 1969 and 1970), 2.1-5.2 m in the late winter of 1972, and 5.7-7.5 m in the late winter of 1975 (Edmondson, 1977). Summer phytoplankton peaks, as measured by chlorophyll a content, increased more than 10 times between 1950 and 1964. Oxygen production, a measure of primary production, increased from 4.48 g/m^2 in 1957 to a high of 6.95 g/m^2 in June, 1966 (Edmondson, 1969). Oscillatoria

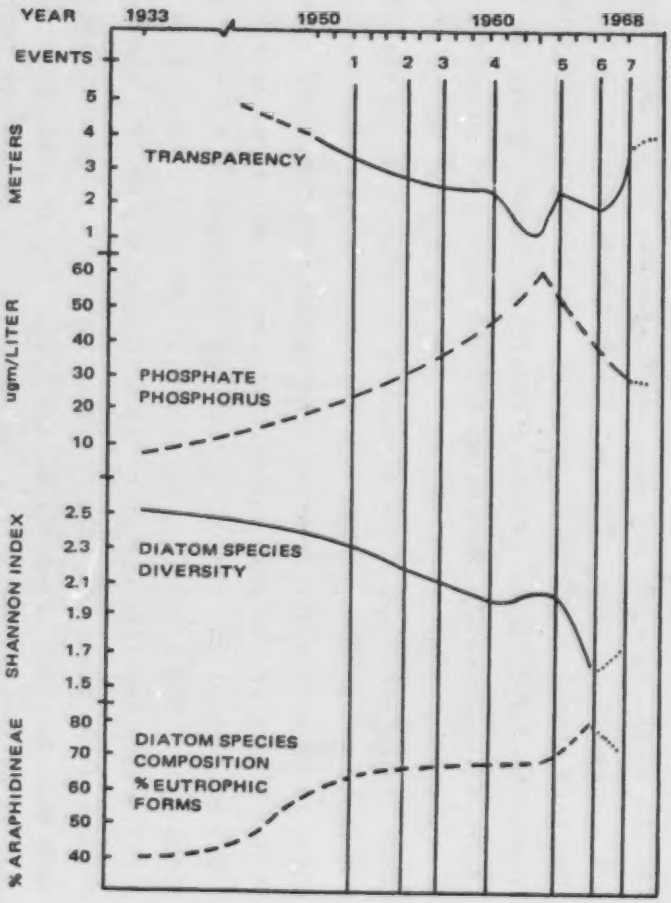

FIGURE 1. REVERSAL OF CULTURAL EU–TROPHICATION IN LAKE WASHINGTON, SEATTLE. NUMBERED EVENTS ARE AS FOLLOWS: 1. ELEVEN SEPARATE SEWAGE PLANTS DISCHARGING TREATED SEWAGE INTO THE LAKE. 2. FIRST NOTICEABLE BLOOM OF NUISANCE ALGAE (OSCILLATORIA). 3. FIRST DETECTED OXYGEN DEPLETION IN BOTTOM WATER (HYPOLIMNION) DURING THE SUMMER. 4. METRO GOVERNMENT SEWAGE PROJECT LEGIS–LATION PASSED (1960). 5. FIRST STEP IN SEWAGE DIVERSION (1963). 6. SECOND STEP IN SEWAGE DIVERSION (1965). 7. ALL SEWAGE DIVERTED (1967). THE TRENDS IN FOUR WATER QUALITY INDICES (TWO PHYSICAL AND TWO BIOLOGICAL) ARE SHOWN.

(ODUM, 1971). (GRAPH BASED ON EDMONSON, 1968, AND STOCKNER AND BENSON, 1967).

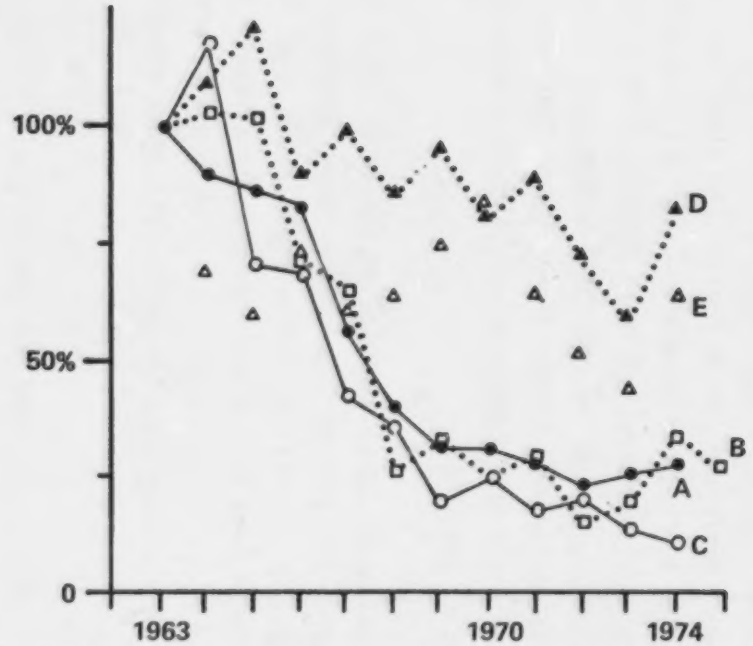

FIGURE 2. RECOVERY OF LAKE WASHINGTON. MEAN VALUES FOR TOP TEN METERS. THE VALUE FOR 1963 (IN PARENTHESES) IS SHOWN AS 100%. A=TOTAL PHOSPHORUS FOR WHOLE YEAR (PERCHLORIC ACID DIGESTION; 65.7 µg/l). B=DISSOLVED INORGANIC PHOSPHATE PHOSPHORUS, JANUARY–MARCH (55.3 µg/l). C=CHLOROPHYLL, JULY–AUGUST (34.8 µg/l). D=NITRATE NITROGEN, JANUARY–MARCH (423 µg/l). E=CARBON DIOXIDE, JANUARY–MARCH (4.05 mg/l). NOTE THAT THE WINTER VALUES ARE FOR A SLIGHTLY DIFFERENT TIME FROM THOSE PUBLISHED EARLIER (EDMONDSON, 1970). SEWAGE DIVERSION STARTED IN 1963 AND ENDED IN 1968. WINTER PHOSPHATE PHOSPHORUS CAN EXCEED THE ANNUAL MEAN FOR TOTAL PHOSPHORUS WHEN THE LATTER DECREASES DURING THE SUMMER AS IN 1974 (EDMONDSON, 1977).

dominated in the mid-1960s, but has been seen only rarely since 1972 (Edmondson, 1977).

Wastewaters were also diverted from the lower Madison, Wisconsin, lakes (Lake Waubesa and Lake Kegonsa). Sonzogni et al. (1975) summarized the data available before and to 15 years after diversion. Soluble phosphorus levels in winter were above 0.3 mg/l P for the years measured before and immediately after diversion in 1958. By 1960, soluble phosphorus levels were below 0.2 mg/l and in 1972 below 0.05 mg/l. Other nutrients also decreased during this period. Algal communities were dominated (99 percent of the algae) by the blue-green alga Microcystis just before diversion (1955-57). Shortly after diversion, Microcystis decreased to 25 percent to 90 percent of the total number of algae, and the community became more diverse. Although the lakes still experience summer algal blooms, they are composed of non-blue-green algae that generally do not form an obnoxious scum along the lake surface and shore.

The situation in lakes the size of the Great Lakes is complicated by the multitude of perturbations on the lakes' ecosystem. Overfishing, introduced species, pollution, and eutrophication have, in combination, had remarkable effects on the biota of the Great Lakes. While it is sometimes difficult to distinguish the effects of individual perturbations, some of the lakes' responses are those which typically accompany increased nutrient loadings.

Hartman (1972) summarized the changes occurring in Lake Erie over the past 50 years. Cultural eutrophication has been the direct cause of changes in the density and diversity of the phytoplankton community and the depletion of oxygen levels. It has also been a significant contributing factor in the changes occurring in the composition of fish and benthic species. The western basin of Lake Erie has been particularly affected by increased nutrient loadings, due to its shallow depth and proximity to the major urban and

industrial centers of Detroit, Michigan and Toledo, Ohio. This basin has been considered an important spawning and nursery ground for fish since it contains many shoals and rocky reefs. Nutrient loadings have increased in the lake over the past 50 years. Although the data are scanty, Beeton (1969) estimated that total nitrogen and phosphorus concentrations had increased by factors of 3 and 2 respectively between 1930 and 1958. More recently, a five-fold increase in the soluble phosphorus concentrations of the lake was reported between 1950 and 1961-62 (Verduin, 1969) and a 50 percent increase between 1961-62 and 1967-68 (FWPCA data cited by Verduin, 1969).

Phytoplankton have responded directly to the increased nutrient loadings with changes in species composition and increased biomass. Between 1928 and 1958, the plankton of the western basin of Lake Erie was dominated by diatoms. By 1969, increasing numbers of flagellates, blue-green and green algae as well as nuisance growths of Cladophora were reported during the summer and fall (Anon., 1969). Cladophora is a filamentous alga which attaches to lake bottoms near shore and to depths of 5 m. Wave action and currents frequently break the alga loose, forming dense floating mats which, when they sink and decompose, add significantly to the biochemical oxygen demand of the sediments. Frequently the mats wash ashore and produce foul odors and attract insect pests as they decompose. Davis (1964) reported that the density of phytoplankton sampled at a water filtration plant near Cleveland, Ohio, increased from an average of about 400 cells/ml in 1920-1937 to 900 cells/ml in 1944-48 and to 1500 cells/ml in 1956-63. In the late 1960s and early 1970s the length of the spring and fall algal abundance maxima were noticed to have increased considerably and the summer and winter minima had become less pronounced. Hartman (1972) cited the work of Davis, who studied the changes in plankton species occurring in the central basin of the lake. Between 1920 and 1949, the spring phytoplankton maxima were usually dominated by the oligotrophic diatom Asterionella; since then, Melosira, a diatom typical of more nutrient-rich waters, has been the dominant form.

The fall pulses have shown a progression from Synedra and Melosira before 1947, to the mesotrophic diatom Fragilaria, the green alga Pediastrum, and the eutrophic blue-green algae Anabaena and Aphanizomenon. As recently as 1970, the fall pulses were dominated by a succession of blue-green algae, Anabaena, Microcystis and Aphanizomenon (Hartman, 1972).

Summer oxygen depletion has been severe in parts of Lake Erie. The western basin usually remains unstratified throughout the summer, owing to the mixing action of winds and currents in this shallow basin. Oxygen depletion in the hypolimnion of the central basin has increased in severity. While first noticed in 1930 as an isolated area, the zone of oxygen depletion had increased to 3,600 km^2 in 1959, 6,600 km^2 in 1970 and peaked in 1973 at 11,600 km^2. In 1973, the anoxic region covered 94 percent of the hypolimnion and 70 percent of the entire basin (Great Lakes Water Quality Board, 1976). Moderately low oxygen levels (2-5 mg/l) were reported in the deep eastern basin as well (Hartman, 1972).

Oxygen depletion, a result of increased nutrient loadings, has presumably been responsible for the devastation of the benthos of Lake Erie. Before 1953, the benthos of the western basin was dominated by nymphs of the mayflies ("Canadian soldiers") Hexagenia rigida and H. limbata occulata before 1953. During the summer of 1953 populations were nearly exterminated during a period of oxygen depletion which resulted after 28 days of hot, calm weather. By 1961, Hexagenia populations had decreased in size to less than 1 percent of their former abundance (Carr and Hiltunen, 1965). The period from 1930 to 1961 saw an increase in species typical of eutrophic, oxygen-poor waters in the western basin; sludge worms, midge larvae (Chironomus) and tubificids (Limmodrilus) became dominant in the benthos (Beeton, 1969; Verduin, 1969; Carr and Hiltunen, 1965; Veal and Osmond, 1968).

Oxygen depletion has been a factor contributing to the enormous change in fish populations in the lake over the past 60 years; other perturbations, including intensive exploitation, temperature increases, construction of dams, land drainage, introduced species, and toxic pollutants were also important. Populations of lake trout, lake herring, sturgeon, whitefish and blue pike were intensively overfished, causing a rapid decline in abundance; however, decreased oxygen levels have apparently precluded the recovery of these populations (Hartman, 1972). Increased algal growth and deposition of organic sediments on the lake bottom are suspected as factors in the decline of species such as walleye, blue pike, sauger, lake herring, and whitefish that require rocky gravel shoals for spawning (Beeton, 1969). A more complete discussion of the change in the commercial fishery of Lake Erie is contained in Section V.

Eutrophication of Lake Erie has had little effect on species such as gizzard shad, goldfish, channel catfish, emerald shiners, brown bullhead, carp, and white bass. These species spawn at shallow depths, usually less than 5 feet, and have short incubation periods (less than 5 days). These two factors tend to minimize the effects of low oxygen levels that are detrimental to cold water bottom spawners. The freshwater drum is one fish that has capitalized on the reduction of competition and predation resulting from eutrophication and other stresses. Since 1950, survival of the young-of-the-year has increased. Unfortunately, freshwater drum has almost no value to commercial fishermen and sports anglers (Hartman, 1972).

General Effects on Aquatic Plants, Zooplankton and the Food Chain

As seen in the preceding examples, increased algal biomass and productivity are the typical responses to increased nutrient loading, but the specific biotic changes which accompany eutrophication are variable, complex, and difficult to predict. The changes in the phytoplankton community normally occurring during cultural eutrophication were summarized by Prescott (1968). The cyanophytes (Aphanizomenon, Microcystis, and Anabaena) typically dominate the plankton, during late summer and early fall blooms, and diatoms, such as Asterionella, Fragilaria, Tabellaria, and Meridion, also become more abundant, especially in the spring. Chlorophyte genera such as Oocystis, Treubaria, Golenkinia, Tetrallantos and Ermosphaera, characteristic of oligotrophic lakes, are replaced in eutrophic waters by Pediastrum, Botryococcus braunii, Coelastrum microporum and Scenedesmus spp. These generalizations are supported, in part, by the previously cited studies of Schindler (1971), Schindler et al. (1971), Langford (1948), Smith (1969), and Vanderhoef et al. (1974).

In general, phytoplankton community diversity decreases as lakes become more eutrophic. Community diversity (H') is commonly measured by the Shannon-Wiener method which incorporates the number of species present as well as the distribution of individuals among species (i.e., evenness). Phytoplankton diversity is significantly correlated to the evenness component but not to the number of species. Oligotrophic lakes, although low in productivity, are more diverse than eutrophic lakes. Poor nutrient conditions favor no one species, and overproduction of algae does not occur. Eutrophic lakes, on the other hand, tend to become dominated by one or two algal species. Other species may be reduced in number by adverse environmental conditions, or by biological interactions such as competition, grazing, parasitism, and antibiosis (Porter, 1977).

High phytoplankton production is sometimes associated with a decrease in the production of aquatic macrophytes. Jupp and Spence (1977), for example, noticed that Anabaena spp. were particularly effective in reducing the amount of light available for Potamogeton filiformis and other macrophytes in Lock Leven. They suggested that high phosphorus levels (0.05 mg/l) in the lake during the summer months were responsible for the dense phytoplankton growths, which in turn, may have been responsible for the reduction in macrophyte biomass. Light attenuation and/or pH increase were thought to be the causative factors.

Changes in the zooplankton during eutrophication are not as well documented as phytoplankton changes. Macroconsumers (Calanoida and larger Cladocera) dominate in oligotrophic lakes where nannoplankton (less than 50u) are in the majority. These grazers are filter feeders, which consume large numbers of diatoms, small green algae, and nanno-flagellates. The colonial and filamentous cyanophytes and green algae ("net phytoplankton"), characteristic of eutrophic waters, have defenses against zooplankton grazing; e.g., clogging of filter feeding apparatus. The zooplankton inhabiting these waters are usually microconsumers, utilizing algae indirectly by the way of the associated bacteria and detritus (Porter, 1977 and Pederson et al., 1976). Smith (1969) and Gentile (1971) reported a decrease in the number of zooplankton during blooms of blue-green algae. A greater proportion of algal productivity is used for food in oligotrophic lakes, and the zooplankton food supply is used less efficiently as the trophic state advances towards eutrophy (Hillbricht-Ilkowska, 1972). Welch et al. (1975) showed that net plankton was utilized less than 10 percent as much as the smaller plankton.

These observations are of major importance in studying the impact of eutrophication on the food chain in eutrophic waters. An increase in phytoplankton primary production is not necessarily reflected

in a proportionate increase in zooplankton secondary productivity. Often the increased phytoplankton biomass is not available to higher trophic levels because blue-green algae are not utilized by herbivores. Increased blue-green algal growth provides an energy source to the decomposers or to a different component of the secondary trophic level (e.g., zoobenthos).

Changes in the composition of higher trophic levels can be expected to occur as the result of changes in the phytoplankton community. Bottomfeeding fish are able to capitalize on their increasing food stocks. In this manner, the trophic structure of nutrient-enriched waters has completely changed from the phytoplankton-zooplankton-carnivorous fish food chain of oligotrophic waters (Pederson et al., 1976).

Toxic Effects of Blue-Green Algae

Those species which thrive in eutrophic waters are endowed with mechanisms enabling them to prosper, sometimes at the expense of other species. Some blue-green algae typical of eutrophic waters, e.g., Anabaena, are able to fix atmospheric nitrogen. When dissolved nitrogen becomes scarce relative to other nutrients (the case with sewage-enriched waters), the growth of other algae may be limited; the blue-greens are able to proliferate by deriving nitrogen from the atmosphere. Shapiro (1973) has also found that CO_2 and phosphorus uptake kinetics favor the dominance of blue-green algae over green algae.

Blue-green algae appear to influence phytoplankton species as well as other levels of the food chain. Keating (1977) found that blue-green algae from a eutrophic pond produce extracellular metabolites which influence the succession of algal species. Macronutrients and light determine the level of productivity and, to some extent, major dominant groups; algae are adapted for optimal growth at

specific nutrient concentrations and light intensity (Vollenweider, 1968). Other factors, including micronutrients and temperature patterns, determine which species will dominate. Extracellular metabolites are thought to influence the seasonal succession of dominant organisms within a eutrophic lake. Keating found that cell-free, axenic filtrates of each dominant blue-green alga produced only positive or neutral effects on the growth of the succeeding dominant organism and only negative effects on the growth of its immediate predecessor. These results suggest a very novel method which may be of use, after further development, in the management of the nuisance blue-green algal blooms typical of eutrophic water.

Earlier studies also showed the influence of some blue-green algal metabolites on other phytoplankton groups. Gentile (1971) summarized the earlier reports which showed that waters containing blooms of Microcystis, Oscillatoria, Anabaena inhibited the growth of other test algae.

While some blue-greens may indirectly impact herbivores by reducing their food resources, some also directly influence other trophic levels by the production of toxins. The toxin may be produced by the alga, or by the bacteria which appear in association with the alga, or released by the alga upon decay (Gorham, 1960). Gentile and Maloney (1969) isolated a toxic strain of Aphanizomenon flos-aquae from Kezar Lake, New Hamphire. They used bioassays to investigate the effects of the toxin from lysed cells on several zooplankters. Bosminia longirastris would be expected to be killed by a bloom containing 1×10^6 algal cells/ml, while elimination of Daphnia catawba would require 4×10^7 cells/ml (Gentile, 1971). Daphnia catawba could not be cultured with healthy cells of this algal even when adequate food sources were provided. Gentile (1971) supposed that Daphnia ingestion of the small filaments of the toxic algae was responsible. Prescott (1968) reported on an extrametabolite of

Microcystis sp., another typical eutrophic cyanophyte, which inhibits the growth of Daphnia and Cyclops. Mills and Wyatt (1974) found that some strains of blue-green algae were toxic to the ostracod Cyprinotus incongruens, which typically inhabits the benthic habitat of inland littoral zones. Additional evidence exists to support claims of toxicity of some blue-green algae to zooplankton (Gentile, 1971).

Fish kills in eutrophic waters have been attributed to toxins as well as oxygen depletion produced by blue-green algal blooms. Sawyer et al. (1968) reported on massive fish kills occurring after copper had been used to control blooms of Aphanizomenon flos-aquae in Lake Winnesquam and Kezar Lake, New Hampshire. Toxin from the killed algal cells was thought to be the primary cause of the fish mortality, although copper may have acted synergistically to enhance sensitivity to the toxin. In laboratory assays algal extracts were toxic to mummichogs (Fundulus heteroclitus) and sheepshead minnows (Cyprinodon variegatus) by intraperitoneal injections (LD_{100} = 0.5 mg algal extract/kg). Golden shiners (Notemigonus crysoleucas) were sensitive to concentrations of the toxin which would normally occur during a bloom (Gentile and Maloney, 1969). Sawyer et al. (1968) demonstrated the toxicity of a natural population of Ap. flos-aquae from Kezar Lake to sunfish (Lepomis gibbosus), guppies (Lebistes reticulatus) and white suckers (Catostomus commersoni), despite high (> 6 mg/l) concentrations of dissolved oxygen.

Two common species of Microcystis produce extrametabolites toxic to Gambusia, the mosquito fish (Prescott, 1968). Carmichael and Gorham (1977) and Carmichael et al. (1975) reported that goldfish were susceptible to a toxic strain of Anabaena flos-aquae if the toxin was ingested. The toxin was apparently not able to enter the fish through the gill membrane. Table 1 lists other algal species thought to be toxic to some fish species and Schwimmer and Schwimmer (1968) summarize the literature from the late 1800's.

Terrestrial animals, including livestock and water fowl, have been poisoned by drinking water containing high concentration of some blue-green algae. While toxic blooms have been reported from most regions of North America, they appear to be of particular importance in eutrophic lakes and sloughs in Western Canada (Carmichael et al., 1977). Anabaena flos-aquae, Aphanizomenon flos-aquae, and Microcystis spp. have often been implicated in dramatic and serious poisonings (Gorham, 1964a). In the fall of 1952, Storm Lake, Iowa, experienced a series of toxic blooms of Anabaena flos-aquae. The algal poison was blamed for the deaths of 5000 to 7000 Franklin's gulls, 560 ducks, 400 coots, 200 pheasants, 2 hawks, 50 fox squirrels, 18 muskrats, 1 skunk, 1 mink, 15 dogs, 4 cats, and 2 hogs (Rose, 1953). Gorham (1964b) reported than an endotoxin from a strain of Microcystis aeruginosa was toxic in laboratory experiments to sheep, rabbits, and calves, but ducks were resistant. Carmichael and Gorham (1977) and others (Carmichael et al., 1975) have reported on toxicity tests using toxic strains of Anabaena flos-aquae on pheasants, mallards, calves, and laboratory rats and mice. Small amounts (18-1200 ml) of a 20 mg dry wt/ml (minimum lethal dose in a mouse by intraperitoneal injection=60 ml) algal suspension was toxic to most species in a short time (several minutes to several hours). In another experiment, Carmichael et al. (1977) concluded that the toxin from one toxic bloom of Anabaena flos-aquae paralyzed the respiratory muscles of calves and probably other livestock. They estimated that a 60 kg calf consuming 1.2 l of a concentrated bloom of toxic A. flos-aquae (Type a, strain NRC-44-1) would die. Mackenthun and Ingram (1967) stated that, rather than death, Microcystis may produce illness, characterized by decreased milk production in cattle. Table 1 lists the algal species most often implicated in livestock poisonings.

Most blooms are not directly toxic; of 11 species of blue-green algae which dominate eutrophic blooms, Gorham (1964a) found only two toxic species. Genetic factors determine the potential for toxin production (Gentile, 1971). Environmental factors, such as light

Table 1. Death-iducing algae (after Prescott, 1968).

Algal Species	Ducks Birds	Sheep	Cattle	Fish	White Mice
Anabaena flos-aquae	+	+	+		+
Anabaena circinalis				+	
Anabaena limnetica					+
Anabaena Scheremetievi					+
Anabaena inaequalis				+	
Anabaena Nadsonii				+	
Aphanizomenon flos-aquae	+		+	+	+
Aphanothece nidulans					+
Aphanothece cyanea					+
Gloeotrichia echinulata					+
Lyngbya Birgei					+
Microcystis toxica			+	+	
Microcystis aeruginosa		+	+	+	+
Nodularia spumigena		+	+		+
Scenedesmus sp					+
Chorella sp					+
Gonyaulax catenella*				+	
Cymodimium brevis* and other spp				+	

*Marine species, blooms most often related to availability of vitamins.

intensity, temperature, pH, and available inorganic nutrients govern the dominance of the species and strain of algae as well as the production of toxin, if any. In addition, the stage in the life cycle of the algae (e.g., spore production) governs when the toxin is produced and, possibly, released (Gorham, 1964a).

Carmichael and Gorham (1977) produced evidence demonstrating that biological factors, i.e., the presence of certain toxin-depressing bacteria, influence the toxicity of Anabaena flos-aquae cultured in the laboratory. The relative abundance of toxic and nontoxic strains of the alga was another factor. They estimated that for an Anabaena bloom to be toxic to livestock, it should consist predominantly of A. flos-aquae and be composed of 80-90 percent toxic filaments. For a bloom to be toxic, the toxin must be present in sufficient quantity and in a suitable form, as well as consumed by susceptible species and in sufficient amounts for deaths to occur. Dilution, adsorption, or destruction of the toxin may render it harmless before effects on aquatic or terrestrial species are felt (Gorham, 1964a). These variables make accurate predictions of the toxicity of algal blooms difficult.

It can be concluded, from the foregoing studies, that blue-green algae common in eutrophic lakes and ponds can be toxic to other members of the aquatic and terrestrial community. The nature of this interaction is variable and needs more study (Gentile, 1971). It is difficult to adequately assess the hazard of toxic algae, as it relates to eutrophication. It has been established that several algal species, common in eutrophic lakes and ponds can be toxic to some animals. From the evidence available, it would appear that toxic algae in eutrophic waters are an infrequent problem. Animal poisonings from toxic algae occur infrequently, according to Schwimmer and Schwimmer (1968) who found only 65 episodes reported in the literature between 1878 and 1962, although they appear to be a periodic problem in Alberta and

Saskatchewan (Carmichael et al., 1977). On the basis of reported frequency, toxic blooms appear to be of slight significance. On the other hand, some toxic blooms may go unreported in the scientific literature because of the difficulty in culturing the algae, identifying causative agent, and obtaining a sample of the transient toxic bloom. In addition, animal losses are more easily attributed, by the farmer, veterinarian or pollution biologist to better known and better understood causes (Schwimmer and Schwimmer, 1968). On the basis of the evidence available, it is difficult to classify the significance of toxic blooms as anything but slight. However, we should be aware that as the frequency and extent of algal blooms increases with accelerated eutrophication, the frequency of occurrence of a toxic strain of the cyanophytes typical of such blooms may also be increased (Gentile, 1971). Carmichael and Gorham (1977) believe that since cyanophyte blooms are potentially toxic all blooms are suspect, even though most may actually be nontoxic.

Effects on Fish

Eutrophication is accompanied by changes in fish communities as well. Moderate enrichment of lakes and ponds is a standard management technique in some parts of the country, designed to increase the production of favorable game species. However, with uncontrolled or excessive nutrient enrichment, some fish populations suffer. In 1972, an international symposium was held in Canada to explore the changes in salmonid and associated communities in oligotrophic lakes (SCOL) resulting from the separate and combined impact of eutrophication, exploitation, and introduced species (see J. Fish. Res. Bd. Canada 29:6). A similar symposium was held in 1976 concerning percids (PERCIS) (see J. Fish Res. Bd. Canada 34:10). Colby et al. (1972) summarized the findings in SCOL lakes related to eutrophication and Leach et al. (1977) summarized the work in PERCIS lakes. While there was no SCOL nor PERCIS lake impacted solely by eutrophication, salmonid and percid communities were considered to have certain

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characteristic responses to nutrient enrichment; for salmonids: initial increased growth and increased incidence of parasitism and later, possible introgression, reduction of natural reproduction, and predictable species dominance; for percids: growth and biomass increase to a threshold level followed by decrease, increased parasitism, changes in food habits, distribution and spawning habits.

In several eutrophic SCOL lakes, some salmonid species enjoyed increased growth rates: the coregonines of the Bodensee (Coregonus wartmanni and C. macrophthalamus) and the Schliersee. In the Bodensee, abundance of C. wartmanni (Blaufelchen) increased as a result of eutrophication, because of the increased amount of available food (Numann, 1972). Northcote (1972) reported on similar effects on kokanee (Oncorhynchus nerka) in Kootenay Lake. Percids responded to cultural eutrophication with an increased growth rate in the young and/or adult fish. This was true for walleye, blue-pike and Eurasian perch in North American and European lakes, but additional evidence suggests that at some point in the trophic continuum, the growth rate of percids will peak, then decline (Leach et al., 1977).

The incidence of parasite infestations increased in salmonids and percids in several SCOL and PERCIS lakes, possibly as a result of eutrophication. Nutrient enrichment may have enhanced the growth of the parasite populations by contributing to the greater abundance of macrophytes, necessary for the completion of the life cycle of the snail-fish-bird type parasite. For example, Eurasian perch mortality in the Bodensee was blamed on the parasite Diplostomum valvens, while periodic small year classes of coregonines have been blamed on increased infection by Taenia (Numann, 1972). Rainbow smelt mortalities in Lake Erie have been attributed to the microsporidean Lugea hertwigi, which infests 90 percent of the lake's smelt population (Dechtiar, 1972). The incidence of Lymphocystis sp. infestation of walleye was greater in the more eutrophic sections of Lake of the Woods (Schupp and Macins, 1977).

Introgression may occur if the segregative mechanisms between closely related species are destroyed. It has been suggested that eutrophication may cause introgression by destroying the spawning grounds of one species, causing it to spawn in another area which is also used by a closely related species. Blue pike and walleye hybridization may be a result of this (Regier et al., 1969), but the process is not clearly understood.

Reproduction of many salmonids has been adversely affected by the oxygen depletion and siltation of spawning grounds accompanying eutrophication. At the same time, unaffected species, like perch, have become more abundant due to the reduction in competition or predation. Species that use shoals of lakes or lake bottoms for spawning and whose eggs have long incubation periods are particuarly adversely affected. Christie (1972) suggested that lake trout populations in Lake Ontario suffered from increased growth of Cladophora, a response to nutrient enrichment. Decomposition of Cladophora caused oxygen depletion fatal to overwintering eggs. Since the intensification of eutrophic symptoms, burbot no longer make winter spawning runs to the deepest part of the Bodensee, but spawn in the sublittoral zone (Numann, 1972). Percids in Lake Constance, too, have changed their distribution and spawning habits in response to oxygen deficits, decreased transparency, and organic matter deposition (Hartmann and Numann, 1977). Some evidence exists which suggests that reproduction of coregonines in the Bodensee (Numann, 1972), blue pike in Lake Erie (Smith, 1972c) and whitefish in the Bay of Quinte (Christie, 1972) suffered as a result of oxygen depletion.

The succession of fish species in the Great Lakes was studied by Smith (1972c) who noticed that fish families and species within families declined and disappeared in the same order from different lakes, or regions of lakes, during periods of rapid changes in water

quality. Other influences, e.g., marine species and commercial fish preference, seemed to have no impact on the sequence, indicating a more subtle control of species dominance. The stresses accompanying eutrophication appear to act first on those species close to the southern end of their range and temperature preference. Susceptibility of percid species to the effects of progressive nutrient enrichment depends on the following factors: (a) ability to migrate to suitable habitat when preferred habitat becomes unfavorable (Colby et al. (1972) suggested that eutrophication tends to increase the vulnerability of sedentary discrete stocks to other stresses such as commercial exploitation); (b) ability to switch prey species when preferred food items become unavailable, and significantly, (c) reproductive behavior. Lake-spawning walleye and sauger, which broadcast their eggs on the lake bottom, would be most susceptible to organic matter deposition resulting from nutrient enrichment. Yellow perch, however, lay their eggs in ribbons which float upwards from their attachment on the bottom (or macrophyte), ensuring aeration even when the mud-water interface is anaerobic (Leach et al., 1977).

Smith (1972c) found that salmonines (Atlantic salmon, lake trout) were the first species to suffer, followed by coregonines (lake herring, whitefish), then percids (sauger, blue pike, walleye, yellow perch), eventually followed by centrarchids and cyprinids (Smith, 1972b). Eutrophication is related to the disappearance of lake trout in the southern portions of Saginaw and Green Bays, before the sea lamprey became well established (Colby et al., 1972). Increased yellow perch abundance in Lake Ontario may be related to the proliferation of invertebrate food items associated with Cladophora (Christie, 1972). However, nutrient enrichment may not necessarily lead to an increase in fishing yield because the added nutrients may flow through energy compartments not utilized by harvestable fish species.

Changes in the benthos as a result of eutrophication may have a direct effect on fish populations. Colby et al. (1972) suggested that the sequence in fish populations reflects the ability of the fish to use the changing benthic species, i.e., from Mysis and Pontoporeia to mayflies, to chironomids and oligochastes. Oxygen depletion is suspected to cause the change in benthic invertebrates. The diet of percids apparently reflects the change in benthic species; in several PERCIS lakes, large food items such as Hexagenia and amphipods were replaced by small items like chironomids (Leach et al., 1977) as eutrophication advanced.

Effects in Flowing Waters

The preceding discussion illustrates the typical biotic responses occurring during eutrophication: increased algal production, decreased algal community diversity, reduction in populations of benthic organisms and fish which require high oxygen levels, and radical changes in species composition at all levels of the food chain. These responses imply a severe stress on the aquatic ecosystems affected. It should be emphasized that eutrophication problems are not confined to lakes or freshwater. The magnitude of disturbances associated with a given nutrient loading rate is a function of the rate of flow of the receiving water, as well as (among other factors) the relative proportion of nutrients available and needed for algal growth. Swiftly flowing waters rarely develop eutrophication problems, even though their nutrient load could produce algal blooms in a lake with a long detention time. (Reservoirs on nutrient rich rivers experience eutrophication problems for this reason.) A flow rate above 0.3 m/sec will probably prevent the most intense phytoplankton growth seen in lakes (Gleisberg et al., 1976). The short retention time of flowing waters is not conducive to the proliferation of phytoplankton. Stream enrichment is usually manifested by increased periphyton growth.

Recycling of nutrients from the sediments is less likely in flowing waters than in lakes; the sediments may be periodically scoured from the river during periods of high flow. High sediment loads in flowing waters reduce the depth of light penetration and, consequently, the volume of the euphotic zone.

The slower moving portions of a river or estuary, however, like a lake which receives the nutrient load of a stream or river, may experience algal blooms and other problems associated with nutrient enrichment. Several tributaries of the Chesapeake Bay are exhibiting classical signs of eutrophication, resulting from nutrient enrichment from domestic sewage. The Potomac River near Washington, D. C., and the Back River, which receives sewage wastes from Baltimore, are good examples of slow moving waters subjected to nuisance algal growths resulting from nutrient enrichment (Jaworski et al., 1972a and 1972b; Pritchard, 1972). The Patuxent estuary, while still healthy, doubled in productivity in the 1960s; Cory (1974) anticipated similar problems in this area as well, if present trends continue.

Enrichment problems in the Chesapeake Bay appear to be limited, at the present time, to tributaries. Taft and Taylor (1976) studied phosphorous levels in the open waters of the Bay and found levels below that suggested by Ketchum (1969) as a maximum value for unpolluted coastal waters (2.8 ug at P/l) and close to those obtained in studies conducted in 1938 and 1964.

Estuaries are naturally nutrient rich and highly productive bodies of water. The circulatory pattern within estuaries accounts for their high retention of nutrients and consequent high productivity. Surface layers of less saline waters flow toward the ocean while more dense, saline waters flow generally landward. The nutrients tend to be cycled within the estuary and are moved slowly seaward with ocean water (Cronin and Mansueti, 1971).

Nevertheless, in excess, nutrients can cause problems in estuaries, as well as in lakes and rivers. The symptoms of nutrient enrichment of estuaries are similar to those of lakes (Barlow et al., 1963). First, algae respond to increased nutrient concentration with increased growth. Their respiration and decomposition by bacteria leads to oxygen depletion of the deeper waters. Although some atmospheric oxygen can be incorporated into bottom waters, particularly in shallow estuaries through wind mixing, the increased organic load puts added demand on oxygen levels, particularly during calm and/or warm periods. Decreased oxygen levels stress estuarine organisms, especially the sedentary benthos. Commercially and ecologically important benthic animals, as well as their predators, suffer. The elimination of preferred phytoplankton prey species, resulting from changing nutrient levels, could potentially have a devastating effect on the motile larvae of many species of crustaceans (crabs, shrimp), molluscs (oysters, clams) and fish (Hobbie, 1970). Increasing algal concentrations can create surface scums over portions of the estuary, obnoxious odors, and beach foulings (Schofield and Krutchkoff, 1974; Sawyer, 1965).

In particular, Perkins and Abbot (1972) and Waite and Mitchell (1972) reported that nutrient-rich sewage effluents were responsible for the intense growth of the benthic algae Enteromorpha and Ulva in some coastal plain estuaries. Decompositions of these algae on mud flats may lead to the elimination of oxygen in the surface layer of the mud flat, the production of hydrogen sulfide, and the consequent destruction of several important mollusc and polychaete species. Trent et al. (1976) blame eutrophication and other factors for the reduction in populations of blue crabs, oysters and shrimp in a developed marsh along the Gulf Coast of Texas. When compared to the adjacent undeveloped marsh, the developed area had higher nutrient and phytoplankton concentrations, lower oxygen levels, and smaller populations of important crustaceans and molluscs. Another

factor contributing to oxygen depletion in the developed marsh was the creation of small canals, which prevented large scale mixing of the water with the atmosphere. Copeland and Hobbie (1972) mention that nutrient enrichment from urban, agricultural and industrial sources temporarily ruins the oxygenated habitat used by benthic organisms (e.g., clams, snails, marine worms) in the Pamlico River Estuary, North Carolina. Nitrogen is probably the limiting nutrient in this estuary, but phosphorus additions, they speculate, could lead to dominance of nitrogen-fixing blue-green algae, thus making nitrogen available to other forms, as well. The increasing frequency of algal blooms, as well as occasional incidence of anaerobic conditions in late summer, are the effects of nutrient enrichment in this estuary (Hobbie, 1974). Flemer (1972) warns of similar problems in the Chesapeake Bay if present nutrient loading trends are not reversed.

III. Significance of Phosphorus in Eutrophication

Phosphorus is the key element controlling the rate of eutrophication of most lakes. It is one of about 20 nutrients which is needed for algal growth. Any of the nutrients will become limiting, according to Liebig's law of the minimum, if it is present in the smallest quantity relative to the amount needed for growth. Phosphorus is the limiting nutrient in most freshwater lakes, but other nutrients can also become limiting if phosphorus is over-supplied.

Furthermore, of the nutrients which limit plant productivity, phosphorus is the easiest to control. Large amounts of other nutrients, particularly nitrogen and carbon, can enter lake from the atmosphere or from other natural sources; because anthropogenic sources of phosphorus are so important, phosphorus is easier, relative to other nutrients, to control. Phosphorus controls in

productive systems, where another nutrient is limiting, may reduce productivity if phosphorus loadings can be reduced to the point where it is, in fact, limiting.

Oligotrophic lakes are usually phosphorus-limited. Many sewage enriched lakes are nitrogen limited because greater quantities of phosphorus have been supplied relative to nitrogen, resulting in a decrease in the N:P ratio. Coastal waters are usually nitrogen-limited (Ryther and Dunstan, 1971); estuaries may be limited by phosphorus, nitrogen, or another nutrient, or even light (Thayer, 1974). On rare occasions, carbon, limitation has been demonstrated but only in hypereutrophic systems, or eutrophic lakes with very low alkalinity.

Numerous studies have shown that phosphorus levels have a positive effect on algal growth. The research can be divided into two areas: in situ lake examinations and laboratory bottle tests using lake water.

Miller et al. (1974) conducted algal assays on waters from 49 lakes across the United States and determined the factor limiting algal growth in each. The lakes were chosen to represent a wide variety of water qualities and geographic locations. Additions of nitrogen (1.00 mg nitrogen/l as $NaNO_3$) and phosphorus (0.05 mg phosphorus/l as K_2HPO_4) were made, separately and in combination, to flasks containing a culture of Selenastrum capricornutum Printz and lake water. Additions of the limiting nutrient stimulated algal productivity as measured by dry weight. Phosphorus was limiting in most (72 percent) of the lake waters, nitrogen was limiting in 16 percent), and another unidentified nutrient was the limiting factor in 6 percent. The authors then related the limiting nutrient to the

relative productivity of the lake water sample. On the basis of algal dry weight production in the control assays, the lake waters were divided into four categories: (1) low productivity (0.00-0.10 mg dry wt/l), (2) moderate productivity (0.11-0.80 mg dry wt/l), (3) moderately high productivity (0.81-6.00 mg dry wt/l), and (4) high productivity (6.10-20.00 mg dry wt/l). There was an inverse relationship between the relative productivity and the percentage of lakes in each group which were phosphorus limited; i.e., phosphorus was limiting in 83 percent of the low productivity lake waters, 75 percent of the moderate productivity lake waters, and 50 percent of the high productivity lake waters. These results suggest that algal productivity is associated with phosphorus levels, and nitrogen, or some other nutrient, is more likely to be the limiting factor of lake waters with high concentrations of ortho-phosphate.

Gakstatter et al. (1975) arrived at similar results using water from lakes in the National Eutrophication Survey. Many of the lakes selected in the eastern United States were receiving sewage treatment plant effluent and most (80 percent) were considered to be eutrophic. Algal assays were conducted on waters from the lakes east of the Rocky Mountains to determine the limiting nutrient of each. Phosphorus was limiting in 67 percent of the lakes; nitrogen in 30 percent; and another unidentified nutrient in 3 percent. The authors concur with Miller et al. (1974) that nitrogen is more likely to be the limiting factor in lake waters with high concentrations of phosphorus. Sewage effluent contains much lower N:P ratios (2 to 5:1 by weight) than are normally found in natural waters (15:1 by weight). Municipal effluents containing high concentrations of phosphorus could provide phosphorus in sufficient quantity to make another nutrient, possibly nitrogen, limiting.

Whole lake studies have provided a clear picture of algal growth stimulation by phosphorus loadings. Schindler et al. (1971) studied the result of the addition of a single dose of phosphorus and nitrogen, to a small undisturbed lake (Lake 227) in Ontario. Polyethylene tubes were used to isolate water columns within the lake. The tubes receiving phosphorus, in combination with nitrogen and/or carbon, were found to have the highest phytoplankton standing crop, as measured by chlorophyll a, within 20 days following enrichment. Phosphorus appeared to be the primary stimulus for phytoplankton growth. Nitrogen and carbon may have produced additional algal growth stimulation when an adequate supply of phosphorus was available. Additional experimentation with the addition of only phosphorus and nitrogen to the whole lake indicated that eutrophication was induced despite the naturally low carbon levels.

Langford (1948) and Smith (1969) also studied lakes in the Canadian Precambrian Shield (central and eastern regions). They, too, found that algal growth was limited by phosphorus concentrations. Schindler et al. (1971) hypothesized that phosphorus limits algal growth in most lakes of this geological area as well as in the Great Lakes, which receive over half their drainage from land underlain by Precambrian Shield. Their hypothesis is supported by data showing the similarities in phytoplankton, benthic algae and zooplankton communities of the Great Lakes and small lakes in the Precambrian Shield.

Schindler and Fee (1974) and Schindler (1975) reported on another whole lake experiment conducted in the Experimental Lakes Area of northwestern Ontario (Lake 226). The two basins of the lake were completely separated by a nylon coated vinyl sheet which extended across the narrow portion of the lake. Nutrient additions equivalent to 3.2 g of nitrate nitrogen and 6.1 g sucrose per square meter of surface area per year were made to the epilimnia of both basins

in twenty equal weekly doses. Phosphorus (0.60 g/m^2 yr of phosphate phosphorus) was added concurrently to one of the basins. The basin receiving all three nutrients was covered by a bloom of the blue-green alga Anabaena spiroides within two months. The basin receiving only nitrogen and carbon remained similar to its prefertilization state in terms of phytoplankton standing crop and species composition. Lake 304 (Schindler and Fee, 1974; Schindler, 1975; Schindler, 1977) in the experimental lakes area was also divided into two basins; one was used as a control, and to the other was added phosphorus (0.40 $g/m^2/yr$ as H_3PO_4), nitrogen (5.2 $g/m^2/yr$ as NH_4Cl), and carbon (5.5 $g/m^2/yr$ as sucrose). Phytoplankton responses were similar to lake 227, exhibiting large (three-fold) increases in algal standing crop and domination by green and blue-green algae after two years of fertilization. During the third year, the lake was fertilized only with carbon and nitrogen simulating the removal of phosphorus during tertiary treatment. Chlorophyll levels and algal composition returned to the prefertilization state within one to two years. These two experimetns indicate that phosphorus was the limiting nutrient in the two lakes and efforts to control excess phosphorus loadings to such lakes would appear to prevent eutrophication.

Municipal waste treatment facilities have long been implicated as a source of phosphorus in surface waters. The following research shows that sewage effluent stimulates algal growth; the component in sewage responsible for increased productivity is phosphorus in almost all cases where lakes receive the effluents.

Edmondson has studied the deterioration of water quality of Lake Washing n, Seattle, and its improvement following sewage diversion to Puget Sound. Although sewage effluent contains nutrients other than phosphorus, Edmondson's study (1970) concluded

that summer chlorophyll content in the lake was related to phosphorus concentrations of the previous winter (a time of nutrient maxima in some lake waters), and was not related to nitrate or CO_2 content. While the other nutrients were important, they were supplied in excess of need. Following complete sewage diversion in 1968, the winter phosphorus concentration had decreased to 28 percent of the maximum reached in 1963. The nitrogen concentration decreased to a lesser extent (to 80 percent of that in 1963) because of higher N:P ratios (by weight) in the contributing tributaries (19.9-35.2) than in the sewage effluent (0.52-3.67) (Edmondson, 1969).

Edmondson (1970) concluded that phosphorus was more important than nitrogen, based on surface water content of the two nutrients during the spring growth of phytoplankton. In 1933, before the lake became sewage enriched, nitrate was left over after phosphorus had been nearly depleted by growing phytoplankton populations. The N:P ratio was close to 15 (by weight). In 1962, however, after many years of enrichment with phosphorus-rich sewage effluent, nitrate was exhausted before phosphate, and the N:P ratio was decreased. Following sewage diversion phosphorus, once again, was depleted before nitrate, and the N:P ratio increased (Edmondson, 1969).

Similar observations were made by Michalski et al. (1975) in a study of the effects of phosphorus removal facilities for the sewage treatment plan at Gravenhurst Bay, Ontario. Improvements in mean Secchi disk visibility, chlorophyll a level, N:P ratio and phytoplankton quantity accompanied decreases in the total phosphorus concentrations in the euphotic zone and the total phosphorus loadings (Table 2). Secchi disk visibility and phytoplankton quantity were significantly correlated ($p \leq 0.05$) to mean total phosphorus in the euphotic zone, while a similar relationship with other nutrients (N, SiO_2) could not be made. There were indications that N and SiO_2

Table 2. Mean Value for relevant "trophic indicator parameters" for Gravenhurst Bay during the pre- and post-phosphorus removal years 1969 through 1971 and 1972 through 1973, respectively. Data for total phosphorus, inorganic nitrogen, N: P, chlorophyll a and phytoplankton stocks are based on samples collected through the euphotic zone (Michalski et al., 1975).

	Pre-phosphorus removal conditions 1969/1970/1971	Post-phosphorus removal conditions 1972/1973
Total phosphorus loadings ($g\ P/m^2 \cdot yr$)	2.37	0.40
Secchi disc visibility (m)	2.5	3.2
Total phosphorus (ug P/l)	43.3	33.0
Inorganic nitrogen	70.6	141.0
N:P	11.4	17.1
Chlorophyll a (ug/l)	10.8	7.6
Phytoplankton		
a) Quantity (A.S.U./ml)[1]	2,717	1,424
b) Species	Asterionella formosa Aphanizomenon flos-aquae Anabaena spp.	Asterionella formosa Aphanizomenon flos-aquae Anabaena spp. Synura uvella Dinobryon spp. Chroococcus spp.

[1] Areal Standard Units

were limiting algal growth when phosphorus concentrations were very high, prior to advanced treatment. In further support of phosphorus control of eutrophication in Gravenhurst Bay, when phosphorus removal was temporarily interrupted, water quality problems returned. In late 1975, a new sewage treatment plant was brought into operation; for the first six months of its operation (from November to April) removal efficiency was about 40 percent as compared to 90 percent removal at the old plant. The following June a bloom of Aphanizomenon flos-aquae was present, chlorophyll a concentrations increased to pre-1971 levels and transparency was reduced from 3-5 m in the previous year to less than 1 m. Later in the summer, three months after efficient phosphorus removal was once again operating, the degradation of water quality was reversed (Dillion et al., 1977).

Similarly, Lake Erie's western basin has recently shown a reduction in near-shore phytoplankton densities following a decrease in phosphorus loadings (Nichols et al., 1977). Plankton samples were collected weekly at the Union Water Treatment Plant in Kingsville, Ontario between 1967 and 1975. Phytoplankton biomass had decreased steadily since 1971, so that by 1975 there had been a 42 percent reduction over the mean biomass calculated for 1967-1970. Phosphorus loadings to the lake began to decline in the early 1970s as a result of phosphorus removal at sewage treatment plants, legislation banning phosphate detergents and reductions in industrial discharge. Phosphorus concentrations decreased 40 percent from an average of 62 mg/l in 1967-1970 to 37 mg/l in 1976.

The role of phosphorus in eutrophication is more complicated in flowing waters, particularly estuaries. Flowing waters with a heavy sediment load are often limited by light, rather than a nutrient. Pomeroy et al. (1972) contended that phosphorus limits primary production only in the most sediment-free estuaries; no estuary in the southeastern United States has shown evidence of phosphorus limitation. The upper part of the Cheasapeake Bay is vastly influenced by the Susquehanna River and is phosphorus limited (Pritchard, 1972). A

series of dams and reservoirs along the lower course of the Susquehanna trap much of the sediments it carries, so that the river water is relatively clear at the point where it joins the Bay. Algal blooms, particularly Anacystis sp., have, since 1968, appeared more frequently, lasted longer, and covered more area in the upper bay. The lower portion of the Cheasapeake Bay is probably not phosphorus limited throughout the year; the limiting nutrient may change from nitrogen to phosphorus to a vitamin, depending on the season. Fournier (1966) found that phosphorus limited production in September, trace metals and nitrogen in October, and nitrogen in late March, April and May. The shallow estuaries near Beaufort, North Carolina also appeared to be limited by nitrogen and phosphorus, although nitrogen is probably the primary limiting nutrient (Thayer, 1974). Although Knudson and Belaire (1975) stopped short of defining the limiting nutrient in a Texas bayou (a shallow, tidal stream,), their algal assays confirm the biostimulatory effect of a secondary sewage effluent as compared to tertiary effluent with the phosphorus removed. Their study was in response to repeated fish kills caused by oxygen depletion resulting from the respiration of dense algal blooms. Anoxic conditions could also have resulted during periods of low rainfall and slight mixing of fresh and saline waters. Waite and Mitchell (1972) found that both phosphorus and nitrogen stimulated the growth of the estuarine benthic alga Ulva lactuca.

A controversy erupted a few years ago over whether carbon or phosphorus limited algal growth in freshwater. Because of the vast body of literature accumulated since then which supports phosphorus limitation, it is generally agreed at the present time that phosphorus loadings are the principal causative factor in the overproduction of phytoplankton. In most lake systems, enough carbon and nitrogen can invade from the atmosphere to allow eutrophication to proceed in proportion to phosphorus loadings (Schindler, 1977). The role of carbon and nitrogen in eutrophication is still not completely understood, however.

While phosphates induce eutophication problems in surface waters, their availability in terrestrial systems has helped to increase crop production and, in some cases, can benefit water quality. Fertile land is better able to support the vegetative cover that can reduce soil erosion and nutrient loss from land to water. Nevertheless, agricultural runoff continues to be a major source of phosphorus in surface waters.

IV. Critical Phosphorus Concentrations

Eutrophication is well understood in a qualitative sense; phosphorus content is associated with trophic state, standing crop, and productivity in most fresh waters. Nevertheless, definition of the critical phosphorus concentrations in surface waters has been attempted only recently. Variables including flow rate, depth, turbidity and the concentrations of other nutrients determine the effect that phosphorus will have. The threat of eutrophication resulting from phosphorus is lessened in flowing, turbid, or deep waters, and in waters where phosphorus is not the limiting nutrient.

No national criterion on total phosphorus has been established by EPA to prevent eutrophication (U.S. Environmental Protection Agency, 1976a). Some guidelines are available which define, in a general sense, the levels at which phosphorus is expected to become a problem. Obviously, these guidelines are not steadfast rules for all conditions since there are natural conditions which would require consideration of more or less stringent levels. Insufficient evidence exists to establish universally safe phosphorus concentrations or loads, according to Lee (1977).

Mackenthun (1973, p. 176) suggested that, to prevent nuisance aquatic growth, total phosphorus in flowing waters should not exceed 100 ug/l at any point; 50 ug/l P should not be exceeded at the point where streams enter a lake or reservoir. Vollenweider's (1968, p. 65)

survey of the limnological literature allowed him to suggest a danger level of 20 ug/l total phosphorus in lakes. Sawyer (1947) determined a critical level of inorganic phosphorus (10 ug/l P) at the time of spring overturn that would cause nuisance algal blooms in summer. For estuaries, Ketchum (1969) cited Yentsch's unpublished work which suggested that a value of 2.8 ug-at P/l (86.8 ug P/l) as the appropriate upper limit of unpolluted water. Ketchum's own values of 2.55 ug-at P/l in winter and 1.7 ug-at P/l in summer are based on oxygen supply in photosynthesis and oxygen demand, and are meant to be conservative danger signals in evaluating the eutrophication of estuaries.

Many researchers have attempted to define the impact that various loadings of phorphorus have on water quality. The available models range in complexity from Vollenweider's (1968) single nutrient loading concept to the multi-nutrient simulation models described by Chen et al. (1975). There is some question as to whether or not complex ecosystem models are really required for decisions made on the control of phosphorus loadings (Tapp, 1976). Lake phosphorus models are far better developed than estuarine or stream models. Although the models are still evolving, a cursory review of the more simple lake models is warranted. These particular models have been used to describe the effect of varying phosphorus loadings on a large number of lakes.

Vollenweider (1968) was the first to attempt to define the relationship between phosphorus loading rates and lake trophic status. Using data from North American and European lakes, he initially plotted annual total phosphorus loading ($g/m^2/yr$) against mean depth. He empirically defined permissable loading levels (those which maintain a lake in an oligotrophic state) and dangerous loading levels (those which promote eutrophication) based on the known trophic state (as determined by knowledgable limnologists) and relative loading levels of each of the lakes. As shown in figure 3, deep lakes are capable of assimilating higher loadings within a given trophic state than are shallow lakes.

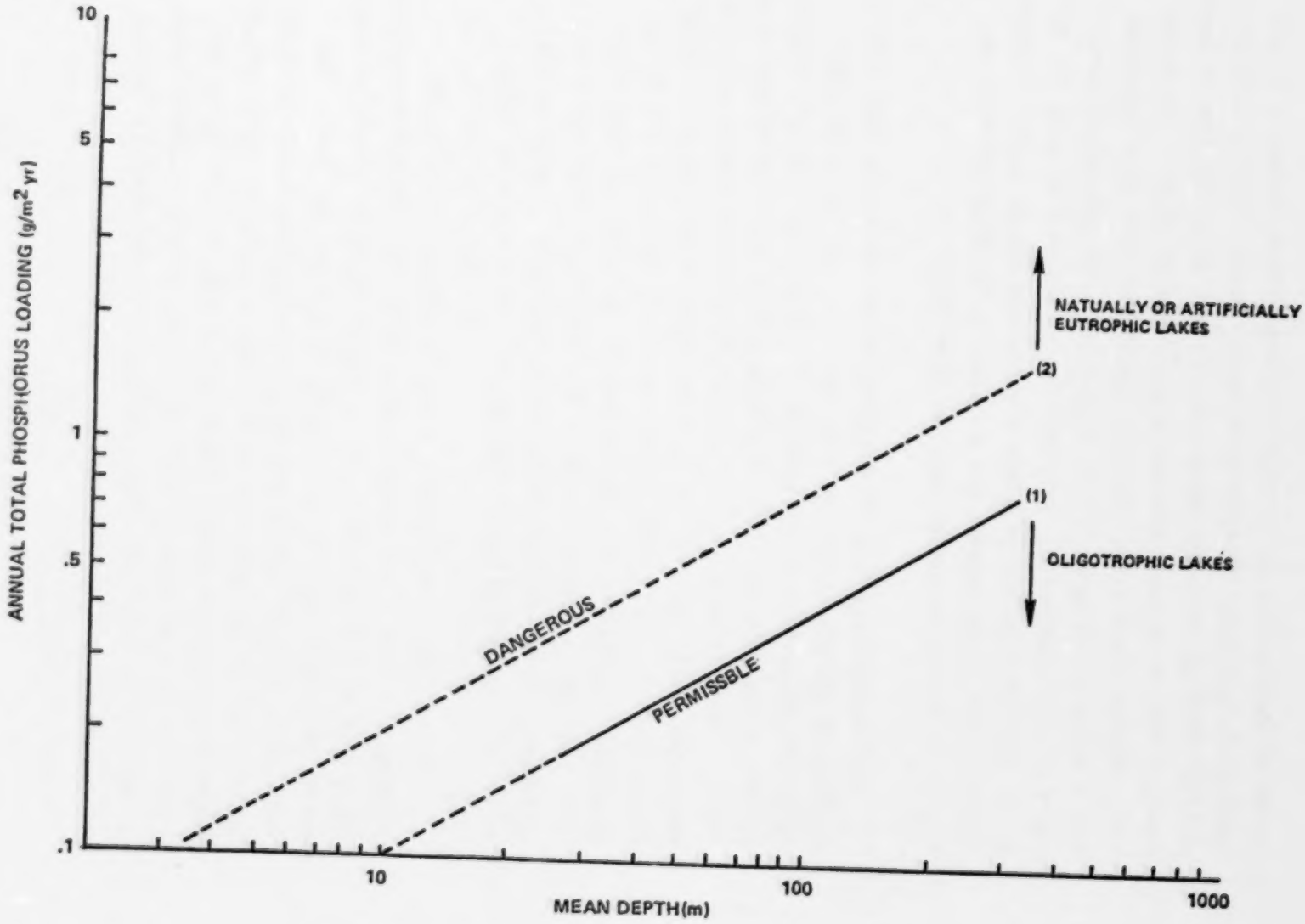

FIGURE 3. VOLLENWEIDER'S (1968) INITIAL PHOSPHORUS LOADING CURVES

More recently Vollenweider (1975) refined his initial model to include hydraulic residence time as well as depth (Figure 4). This model relates annual phosphorus loads to the mean depth divided by the hydraulic residence time. Tolerance loadings were empirically established based on the case where the mean depth divided by the retention time was less than 1. This solution was used to project the tolerance line throughout the commonly occurring ranges of mean depth divided by retention time. The upper line was called the dangerous limit; it was two times the permissible limit, the lower line. With this model, identical loadings may promote eutrophic conditions in lakes with a small mean depth to hydraulic residence time ratio, or oligotrophic conditions in lakes where that ratio is large. Vollenweider (1976) further refined the model using average inflow concentration ($mg/P/m^3$) versus water residence time. Figure 5 illustrates the inflow concentration and residence time associated with certain oligotrophic, mesotrophic and eutrophic lakes and the permissible and dangerous phosphorus loading levels.

Closely related to Vollenweider's latest model is that of Larsen and Mercier (1975, 1976). Their approach is derived from a mass balance model of phosphorus and related the average incoming phosphorus concentration to the phosphorus retention coefficient. Figure 6 illustrates this relationship; it is fitted with lakes of known trophic condition. The upper curve in figure 6 represents the transition from a mesotrophic state to eutrophic; the lower, oligotrophic to mesotrophic. Larsen and Mercier selected in-lake total phosphorus concentrations of 10 and 20 mg/l to establish permissible and dangerous loadings.

Dillon (1975) also used the mass balance approach and defined a relationship between mean depth and a factor derived from the total annual phosphorus loading (L), the phosphorus retention coefficient (R) and the water residence time (ρ). Figure 7 illustrates this relationship along with permissible and dangerous loading rates.

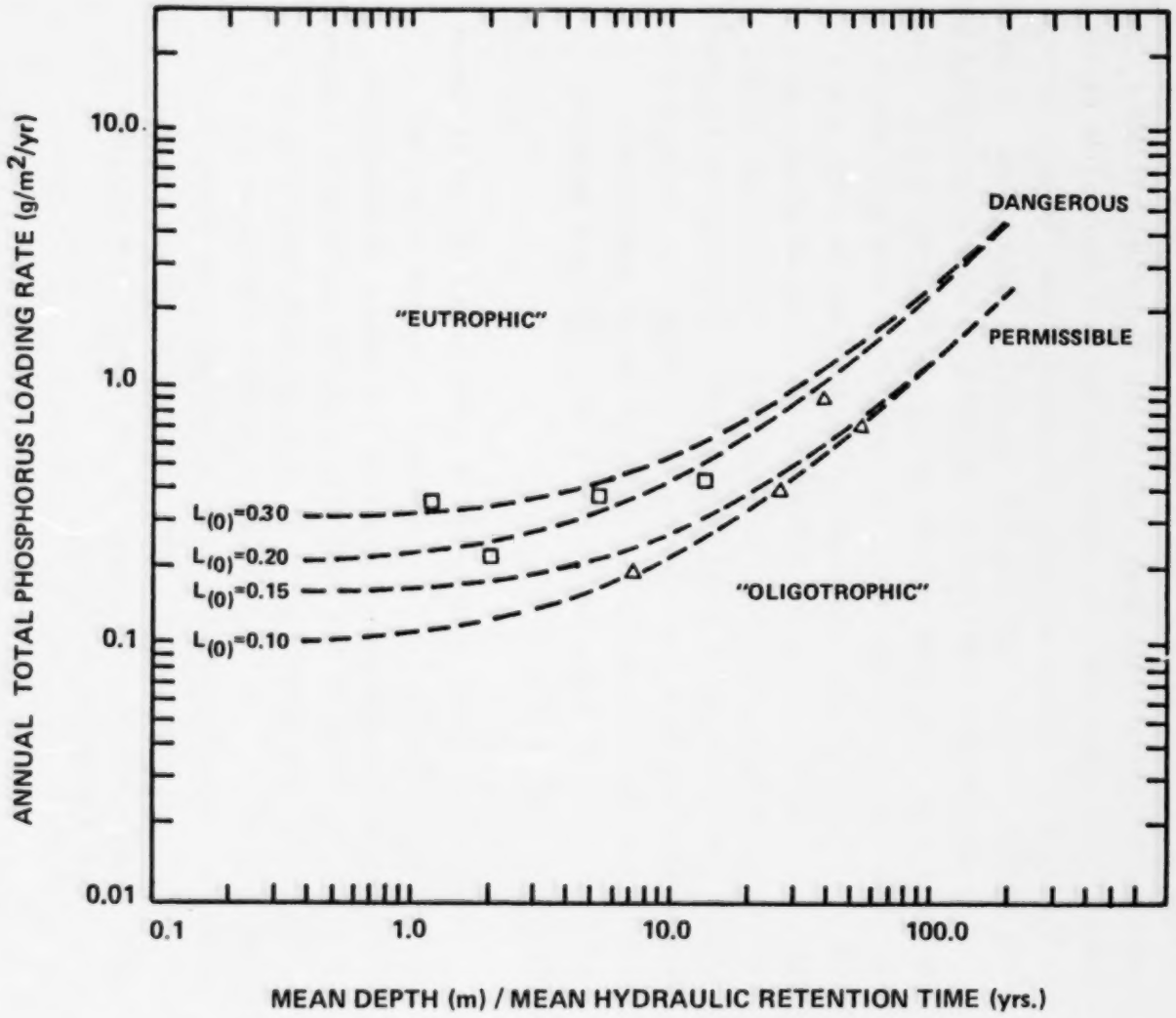

FIGURE 4. VOLLENWEIDER'S (1975) PERMISSIBLE AND DANGEROUS PHOSPHORUS LOADING RATES.

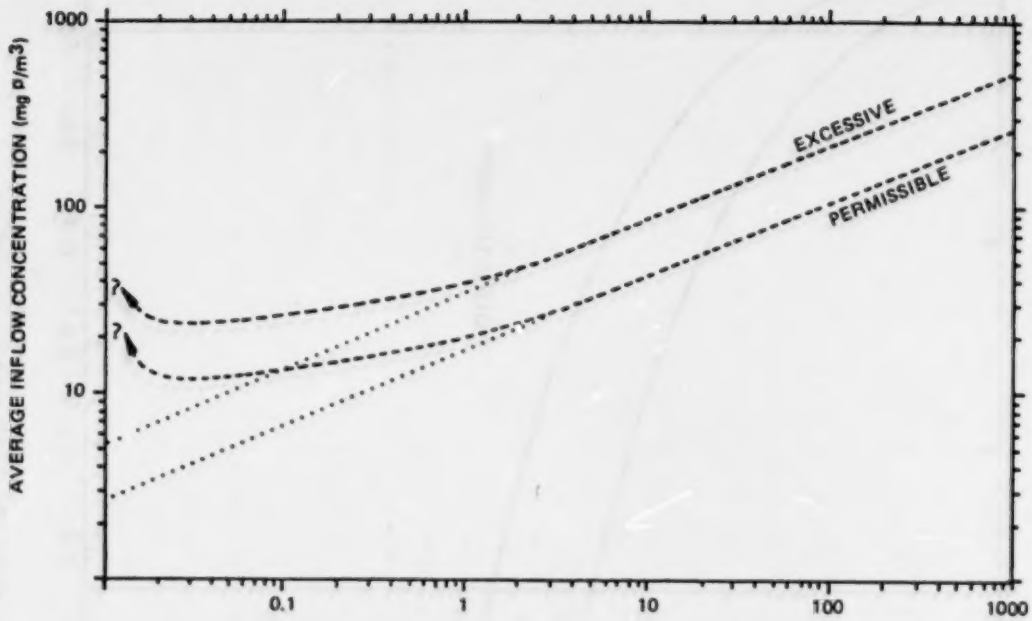

Figure 5. VOLLENWEIDER'S (1976) LATEST PHOSPHORUS LOADING CURVES.

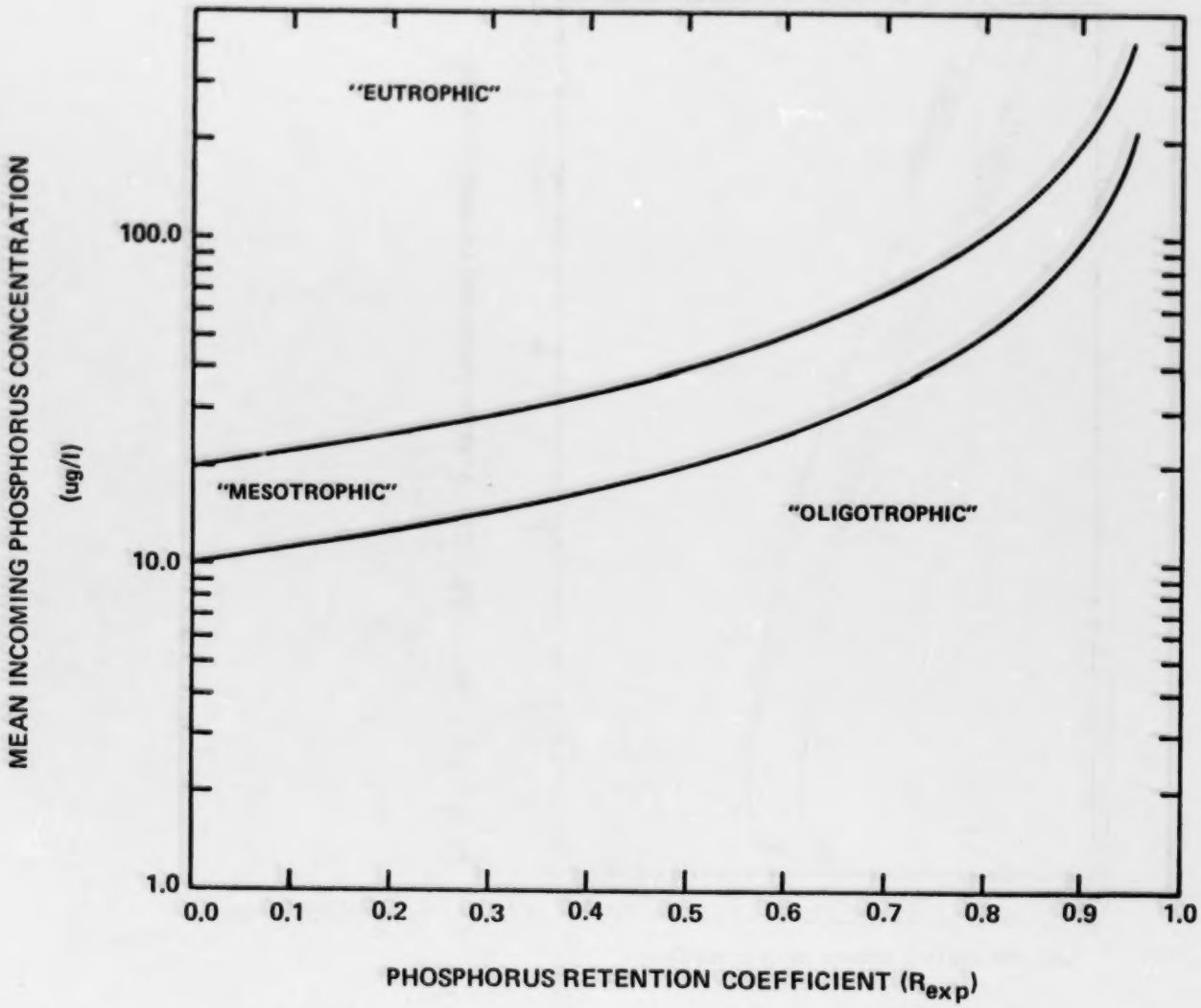

FIGURE 6. LARSEN AND MERCIER'S (1976) RELATIONSHIP DEPICTING CRITICAL PHOSPHORUS LOADING LEVELS. $R_{exp} = 1 - \frac{\text{OUTFLOW LOAD}}{\text{INFLOW LOAD}}$

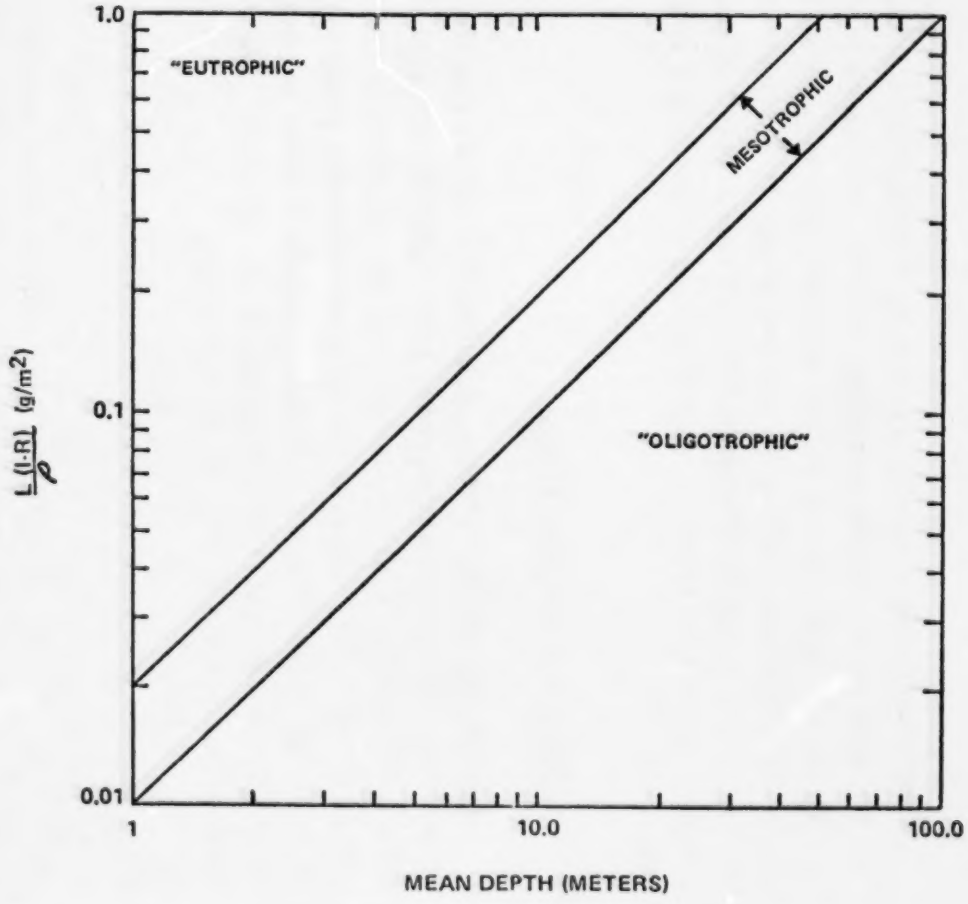

FIGURE 7. DILLON'S (1975) DEFINITION OF CRITICAL PHOSPHORUS LEVELS. L = PHOSPHORUS LOADING; R = PHOSPHORUS RETENTION COEFFICIENT; ρ = HYDRAULIC FLUSHING RATE

Dillion and Rigler (1975) approached the problem of defining critical phosphorus loads from a slightly different point of view. They were able to predict the in-lake spring phosphorus concentration using phosphorus loading, lake morphometry and hydraulic budget information. The spring phosphorus concentration determines the amount of chlorophyll a and transparency. Permissible phosphorus loadings could then be determined, depending on the chlorophyll a concentrations desired in the lake.

Additional simple models, as well as complex simulations, exist for the prediction of safe phosphorus levels. The examples cited here illustrate the major thrust in simple models. It is obvious that the impact of phosphorus loadings on surface waters is difficult to predict universally due to the variability of aquatic systems and the plethora of factors affecting phosphorus availability and ecosystem response.

V. Effects on Human Populations

Besides the radical effects of eutrophication on aquatic communities, human populations using lake resources are also directly affected. Nuisance algal blooms destroy the recreational and aesthetic values of lakes, decrease the value of lakeshore property, and increase the costs of water treatment. Eutrophication has had an impact on the commercial fishing industry in Lake Erie. The following sections contain discussions of these problems.

Water Treatment Problems

Increased algal growth, characteristic of eutrophication, can cause several water treatment problems, including shortened filter runs, objectionable tastes and odors, and occasional severe clogging of intake screens. These problems lead to increased water treatment costs related to the use of greater quantities of chemicals for

adequate treatment, such as alum, lime and activated carbon, and cleaning of water intake screens. A decrease in the volume of water available to consumers, because of clogged intake filters, is also a consideration. According to a 1967 survey of state sanitary engineers, 56% of the total municipal surface water supplies of the United States are affected by water treatment problems associated with eutrophication (Task Group Report, 1967). Silvey et al. (1972) list Anabaena, Aphanizomenon, and Oscillatoria as the species causing most taste and odor problems; they are also often associated with eutrophic conditions.

Water treatment costs are sometimes difficult to relate to water quality. Generally, chemicals for water treatment are used in quantities determined by historical practice or "eye ball" criteria. Monitoring of raw water and the use of appropriate amounts of chemicals for treatment are rare practices (Elly, 1977). An additional consideration, which may mask or eliminate the costs of water treatment, is the plant design. Intake pipes can be designed or situated in areas to avoid the problems associated with eutrophic waters. Several documented cases of water treatment problems and design avoidance associated with eutrophic waters are contained in the literature as follows.

Eutrophication of Lake Erie has been an important cause of increased treatment costs in Cleveland, Ohio. Taste and odor have been perennial problems for many years, requiring the additional use (and cost) of activated carbon and other chemicals. The problems result from increased algal growth causing anoxic conditions in the hypolimnion and the consequent release of some chemicals (iron, manganese) from the sediments (Elly, 1977).

Serious problems occured in the summers of 1966 and 1967 at the Crown Filtration Plant in Cleveland (Potos, 1968). Prevailing southerly winds pushed surface waters to the northern shore of the lake, causing a shift in the thermocline. In the north, the thermocline became deeper while it rose in the south. As a result, the water intake drew from the

hypolimnion instead of the epilimnion. Dissolved oxygen in the hypolimnion was very low, and peak iron, manganese, and phytoplankton concentrations existed. Sediment release of nutrients, iron, and manganese is assumed to have occured under anoxic conditions. Since the hypolimnion was still in the euphotic zone, the phytoplankton responded to the nutrient release. Genera linked to musty taste and odor production were found in water samples: Chlorella, Fragilaria, Melosira, and Mongeotia. The algae and/or mineral content of the water were responsible for the taste and odor problems, while a broken intake, allowing the entry of more turbid water, aggravated the problem. Chlorine demand of the raw water increased three to five times the normal value, due in part to the high concentrations of reduced iron and manganese. But heavy chlorination accentuated the taste and odor problems, requiring heavy doses of alum to correct. Vertical extension of the intake pipe has been recommended (Potos, 1968), while control of the eutrophication problem resulting from excess nutrient loadings has also been advised (Elly, 1977).

In August, 1974, the sand filters at Cleveland's Division Avenue Water Filtration Plant became clogged by massive amounts of the filamentous alga Oedogonium. The shutdown lasted for three days, leaving some of the 400,000 residential, commercial, and industrial users without water. Increased quantities of chlorine, carbon, alum, and permanganate were required immediately before the plant shutdown, while massive amounts of chlorine were used to declog the filters. The probable cause of the bloom of Oedogonium has been attributed, in part, to local nutrient buildup in the eutrophic near-shore waters (Elly, 1977).

An earlier report (Poston and Gamet, 1964) identified several phytoplankton species which cause additional problems in Cleveland. Shortened filter runs, particularly during late winter and early spring, have occurred due to large phytoplankton populations composed primarily of Melosira, and also Tabellaria, Asterionella, Cyclotella and

Stephanodicus. Taste and odor problems are frequently caused by Aphanizomenon.

The overall increase in water treatment costs in Cleveland has risen from $4.50 per million gallons treated in 1968 to $9.00 in 1975. These figures reflect only the treatment costs associated with producing potable water. Costs related to inflation, fluoridation, wages and operating expenses were excluded. According to Elly (1977), the chief engineer has attributed the increase to increased taste and odor problems related to the rate of eutrophication of Lake Erie and changes in the chemicals used and chemical suppliers.

Poston and Gamet (1964) investigated water treatment problems related to algae on Lake Michigan and Lake Erie. They concluded that Tabellaria is the phytoplankton causing the most severe problems with shortened filter runs in Lake Michigan. Taste and odor problems on Lake Michigan were caused primarily by Dinobryon and Uroglenopsis, while Anabaena, a blue-green algae, presented the biggest taste and odor problem in Lake Erie. Activated carbon, although expensive, is used successfully to overcome taste and odor problems in the water supplies of larger municipalities. Treatment problems related to algal growth have been minimized on the Great Lakes by locating the water intakes at depths and sites to avoid such problems.

The incidence of taste and odor problems in drinking water, as well as treatment costs, appears to increase as reservoirs age. Elly (1977) reports that costs at the Lima, Ohio water treatment plant decreased following a change in source from an old reservoir to a new reservoir in 1958. The older reservoir developed algal and taste and odor problems not found in new reservoirs. Prevention of these problems has been attempted by draining unused reservoirs and burning the highly organic bottom sediments.

Gizzard (1976) reported that the 20-year-old Occoquan Reservoir, which serves 500,000 customers in northern Virginia, is exhibiting classic signs of cultural eutrophication. Periodic taste and odor problems arise from the release of metabolic end-products from algae and the decay of organic matter. The onset of taste and odor problems coincides with peak production of the blue-green algae, Anabaena, Microcystis and Aphanizomenon. Large quantities of copper sulfate are used in the reservoir to control algal growth. Usually, alternate water sources are used when reservoirs develop algal blooms (Owens and Wood, 1968).

Water treatment problems relating to Cladophora, a filamentous alga that responds to eutrophic conditions, are few, according to a recent review of the problem (Neil, 1975). Intakes for municipal water supplies are generally located in deep waters, well above the lake bottom, yet at depths where surface accumulations of Cladophora are not a problem; the intakes are designed to avoid the problem. Poston and Gamet (1964), however, reported that Cladophora has caused clogging on intake screens in southern Lake Michigan for several years.

Not only do algae or their products taint water supplies, but some are responsible for causing objectionable tastes in fish flesh. Tabachek and Yurkowski (1976) found that several species of Oscillatoria, a blue-green algae common to eutrophic waters, produced geosmin which is thought to be responsible for a muddy flavor in fish. This was especially damaging to the fish farming industry in the prairie lakes of Manitoba.

Industrial Water Supplies

Reports on the impact of eutrophication on industrial water supplies are uncommon. Taylor et al. (1977) described eutrophication-related problems in the operation of the power plants located on the highly eutrophic man-made cooling lake, Lake Brauning, outside of San

Antonio, Texas. Condenser fouling resulted from the growth of periphytic algae, which is enhanced by eutrophic conditions. Periphytic algae grow and accumulate on the walls of the condenser, decreasing heat transfer efficiency. Shock chlorination is the normal control practice; concentration up to 1 mg/l for 1/2-1 hour are used about 3 times per week. The total annual cost for this treatment is low ($10,077 in 1974), indicating that periphytic fouling is not a serious problem even with highly eutrophic water supplies.

Condenser fouling can also occur as the result of carbonate species deposition, usually in the form of calcite or dolomite. Changes in pH, resulting from the diurnal cycles of respiration and photosynthesis of dense phytoplankton populations, are responsible (Taylor et al., 1977).

Fouling of water intake screens is a common problem in power plants, according to Taylor et al. (1977). Costs of routine maintenance and cleaning generally increase as eutrophication progresses. Nuisance species, such as filamentous algae, carp and shad, are the principle problem. A natural shift in the dominant phytoplankton from the nuisance filamentous Chlorphyceae to Diatomaceae corrected intake screen fouling at Lake Brauning.

In Ontario, Cladophora has been a problem with a number of major generating plants which use surface intakes. At Lakeview, near Toronto, continuing Cladophora problems have required the extension of dikes to ensure a clear water-intake screen. The Ontario Hydro Nuclear Station at Pickering on Lake Ontario also uses a surface intake. Continued clogging of water lines and occasional shut downs, costing $40,000 in lost revenues for each occurence, required the installation of heat exchangers at a cost of $2.7 million. Removal of Cladophora and other trash (up to 5.6 cubic meters) from intake screens is a daily chore during certain seasons (Neil, 1975).

Problems with the treatment of eutrophic lake water for bioler and feed water were reported by Taylor et al. (1977). Removal of organics and suspended solids, associated with phytoplankton growth which interfere with the ion exchange process of demineralization, is the principle problem. In 1975, total costs for the treatment of boiler feed water amounted to $2.14 to $3.13 per 1,000 gal.

Toxic Algae

Some of the algae that respond to eutrophic conditions are toxic to some aquatic and terrestrial animals. A summary of the hazard to domestic animals is contained in Section II. Livestock poisonings are not a common occurrence, even in eutrophic waters, and no data are available on the economic losses associated with toxic algal blooms.

Toxic algae pose little hazard to humans. Most disorders may be classified as gastrointestinal, respiratory, and dermatological. Microcystis, Anabaena and Aphanizomenon are involved most often. Gentile (1971) refers to cases of human intoxication by blue-green algae in the Ohio River basin; Saskatchewan, Canada, and Wisconsin. Gorham (1964b) and Schwimmer and Schwimmer (1968) cited several cases of gastroenteritis associated with involuntary ingestion of toxic blue-green algae. Public health problems associated with contaminated water supplies are considered negligable because any toxins present would be rapidly inactivated during treatment (Gorham, 1964b). For Microcystis and Anabaena flos-aquae, Gorham (1964b) estimated that the minimum lethal oral dose for a 150-lb man is one to two quarts of a thick, smelly, paint-like suspension. He quickly discounted any danger of voluntary ingestion, but recognized a potential problem with involuntary ingestion. According to Gorham, toxic algal blooms apparently can be considered nuisances and perhaps economic hazards, but not public health problems. Gentile (1971) does not refute this, but warns that, with acceleration of eutrophication and increased use of surface waters

for recreation and municipal purposes, the potential for health hazards to result from toxic algal blooms might increase.

Property Values

It is a common assumption that the value of lake shore property is depressed along eutrophic lakes; this assumption is difficult to prove or quantify. Neil (1975) summarized a study by Ormerod (1970) concerning the relationship between algae and real estate values along the north shore of Lake Erie's eastern basin. Ormerod's analysis made use of real estate figures derived from assessment value and multipliers based on the relationship between actual sales dollars and assessed value. His findings show that lake frontage with algae averaged 80 to 85% of the value of clean frontage. Property values declined steeply with the initial accumulation of small amounts of algae, but as algal growth increased, the rate of decline of real estate value leveled off. This study suggested that eutrophication has caused a significant decline in the value of property located over much of the lower Great lakes and perhaps Lake Michigan. If a depreciation of even 10% were applied to lake shore property values, the cost of eutrophication could be considered "staggering".

Dornbusch et al. (1975) described a method to assess the impact of water pollution abatement on property values. Their method can also be used to predict the benefits, in relation to property values, of projected water quality improvements, or the costs of not improving water quality. They compared the property values in 17 areas before and after the correction of water pollution problems. Six of the selected areas had been affected primarily by eutrophication-related problems; reduction of nutrient loadings in each area improved water quality. In each of the selected areas, water quality improvements were associated with increased property values, even when inflation costs were factored in. The amount of increase varied with the type and size of water body, visual and physical access to the water body, distance of the property

from the shore, the orientation of the community toward the water, and the improvements perceived by the community. With comparable improvements in water quality, a greater increase in values was seen with property near large lakes with good or unlimited public access, while a smaller increase was seen in the value of property bordering small rivers with very limited public access.

The six areas investigated with eutrophication problems were Lake Washington, Seattle (2 sites); Green Lake, Seattle; San Diego Bay; Brown's Lake, Wisconsin; and Lake Minnetonka, Minneapolis. Table 3 contains a summary of the problems at each area, abatement procedures, and the property value increase.

In an earlier study, Dornbusch and Barrager (1973) measured pollution abatement benefits for single family waterfront residences at four of the same 17 sites, including one site (San Diego Bay) impacted by excessive nutrient loadings. Sale prices and calibrated local tax assessments were compared before and after pollution abatement. Property values increased from 4.2 to 8.2% after water pollution control in the Bay, depending upon the distance (from 2000 to 100 feet) from the water.

Table 3. Summary of six case histories showing property value increases with water quality improvements. (From Dornbusch et al., 1975)

Study Area	Principal Land Use	Water Quality Problems[1]	Cause of Pollution[2]	Abatement Procedure	Mean Property Value, Increase[3]
Lake Washington I	residential	1,2,3,7	1,4	sewage diversion	13.9
Lake Washington II	residential	1,2,3,7	1,4	sewage diversion	14.7
Green Lake	residential	2,3,4,7	2	dredging dilution	8.0
San Diego Bay	residential	1,2,5,6	1,4,7	sewage diversion	7.0
Brown's Lake	residential	4,5,6,7	1,5,6	sewage weed harvest	17.4
Lake Minnetonka	residential commercial	1,2,3,7	1,4	sewage diversion	4.0

[1]Water Quality Problems: 1. low dissolved oxygen 2. algal blooms 3. decreased water clarity 4. rooted aquatic vegetation 5. declining fish populations 6. high fecal coliform counts 7. phosphorus.

[2]Cause of Pollution: 1. Cultural eutrophication 2. Natural eutrophication 3. Man-made impoundment 4. Nutrients from sewage 5. Nutrients from nonpoint sources 6. septic tanks 7. other

[3]Percentage change in real property values.

Commercial Fishery - The Fish

Phosphate-induced eutrophication has, in some areas, induced changes in fish populations to the extent that populations of commercially valuable fish have been adversely affected. Destruction of oxygen regimes in the hypolimnion is especially detrimental to species of fish and shellfish which are part of the benthic environment, either as adults or embryonic stages.

Radical changes have occurred in the Great Lakes fishery over the past 80 years which can be partially attributed to eutrophication. Other stresses, including over-exploitation, toxic pollutants, chemical contaminants, tributary damming, marsh drainage, siltation, temperature increases and the introduction of new species, have also contributed to the change. Smith (unpublished manuscript) stated that of the factors impacting fish population declines in the Great Lakes, "cultural eutrophication has apparently exerted the environmental stress that has been most closely associated with the terminal decline of many of the species to near extinction or extinction."

The Lake Erie fishery was chosen for an extensive evaluation for several reasons. Not only does an extensive body of data exist to define the extent of eutrophication in the lake (see section II) and the commercial catch, but Lake Erie has traditionally been the most productive, and most diverse fishery of all of the Great Lakes. Annual production from 1915-1965 averaged close to 50 million pounds and often equaled the production of the other four Great Lakes combined, and usually at least one third of the total Great Lakes catch. Apparently the diversity of habitats and the shallow, warm character of the lake have been responsible for the high productivity (Applegate and Van Meter, 1970). It was felt that if eutrophication had any impact on a commercial fishery, the impact would be most obvious with Lake Erie. It should be emphasized that nutrient enrichment of other waters has also had a detrimental effect on commercially important fish and

shellfish (Perkins and Abbott, 1972; Newmann, 1972, Trent et al., 1976; Schneider and Leach, 1977; Ryther, 1954).

The other four Great Lakes have been impacted by nutrient enrichment less than has Lake Erie. Their fishery statistics reflect the important impact that marine invaders (sea lamprey and alewife) have had on important commercial fish species. Populations of these invaders were abundant in Lake Erie because suitable spawning sites were not available for lamprey, and predators were able to check alewife abundance (Smith, 1972a). Nevertheless, Smith (1972a) related deteriorating water quality in Lake Ontario with declines of important commercial species-lake trout, whitefish, lake herring, deepwater ciscos, burbot, and deepwater sculpin, and warned of the similar danger in the deepwater regions of the Upper Great Lakes as well. Smith (1972a) attributes the loss of Lake Erie's valuable species to the vast amounts of waste added to the lake as a result of industrialization. Oxygen depletion in the central basin has made this large portion of the lake uninhabitable for many fish and fish-food species, reducing fishery productivity in this area. Tables 4-6 summarize the change and causes for change in Lake Erie's fishery from the 1800s.

The size of many animal populations is subject to wide, natural fluctuations. Commercial fishery statistics reflect natural variations in fish population size but are subject to market factors as well. They are not necessarily true indicators of abundance. Poundage landed is a more accurate reflection of abundance for the more valuable fish; but fish with little or no value in the market place will not be fished for as heavily. State restrictions and bans resulting from chemical contamination also limit the poundage landed of certain species. Because of natural fluctuations in population size and price fluctuations, it is difficult to accurately relate landings to abundance of even the more valuable fish without a perspective of several years. Analysis of many years of commercial catch statistics can be a valuable tool for measuring the change in fish populations over time. Several good

Table 4. Order of yield of the principal commercial species of fish caught in Lake Erie in selected years from 1908-75[1]. ("Suckers" include both white and redhorse species; catches of "channel catfish" usually include some bullheads before 1952).

Order of yield	1908[2]	1915[3]	1920	1930	1940	1950	1960	1966	1975
1	Lake herring	Blue pike	Lake herring	Blue pike	Blue pike	Blue pike	Yellow perch	Yellow perch	Smelt
2	Blue pike	Lake herring	Blue pike	Yellow perch	Whitefish	Walleye	Smelt	Smelt	Yellow perch
3	Carp	Carp	Carp	Freshwater drum	Yellow perch	Yellow perch	Freshwater drum	Carp	White bass
4	Walleye	Sauger	Sauger	Whitefish	Walleye	Freshwater drum	White Bass	Freshwater drum	Carp
5	Northern pike	Lake whitefish	Yellow perch	Carp	Freshwater drum	Whitefish	Carp	White bass	Freshwater drum
6	Sauger	Yellow perch	Whitefish	Walleye	Carp	Carp	Walleye	Walleye	Channel catfish
7	Yellow perch	Walleye	Freshwater drum	Suckers	Suckers	White bass	Channel catfish	Channel catfish	Walleye
8	Lake whitefish	Freshwater drum	Suckers	Sauger	Lake herring	Channel catfish	Suckers	Suckers	Bullheads
9	Suckers	Suckers	Walleye	Lake herring	Sauger	Lake herring	Goldfish	Goldfish	Quillback[4]
10	-	-	Channel catfish	White bass	White bass	Suckers	Bull head	Bullheads	Sunfish

[1] Adapted from Applegate and Van Meter, 1970.

[2] Ranking of yields of all but lake herring inferred in part from descriptive reports (Canadian landings not reported for majority of species).

[3] Ranking of fourth through ninth species based largely on U.S. records; reports of Canadian landings of sauger, freshwater drum, and suckers lacking.

[4] Sunfish caught in Canadian waters only.

Table 5. Average combined annual United States and Canadian production (thousands of pounds) of major commercial fishes[1] from Lake Erie for specified time periods 1879-1975 (after Hartman, 1973).

Period	Lake Sturgeon	Northern Pike	Lake Herring	Sauger[2]	Whitefish	Blue Pike[3]	Walleye	Yellow Perch	Smelt	Freshwater drum	White Bass[2]	Sucker[2]	Channel catfish[4]	Carp	Other[5]	Total production
1879-1906[6]	1,052	1,356	25,625	3,700	2,402	10,797[7]	--	2,791	--	1,061	611	1,350	604	2,480	100	53,929
1910-1916[6]	77	1,250	27,201	3,656	2,945	9,277	1,756	3,017	--	2,499	383	1,120	1,110	7,544	2,015	63,850
1920-1929[6]	39	77	14,126	2,437	1,675	11,292	1,577	5,356	--	2,367	360	1,090	681	3,189	1,476	45,742
1930-1934	39	62	764	1,943	2,094	14,623	2,113	12,382	--	2,381	447	1,462	700	2,659	1,594	43,263
1935-1939	31	29	1,070	1,414	2,696	18,526	3,515	6,444	--	3,359	655	980	641	2,689	1,964	44,013
1940-1944	22	37	283	878	4,058	13,517	3,779	3,869	--	3,624	553	628	948	2,593	1,744	36,533
1945-1949	25	21	6,067	567	4,701	12,509	5,807	4,245	--	3,965	701	506	1,093	2,077	2,226	44,510
1950-1954	14	12	475	354	2,297	13,535	7,566	6,784	890	3,492	3,485	661	1,589	3,077	973	45,134
1955-1959	14	14	128	21	749	10,078	10,267	19,540	4,345	4,020	5,092	413	1,770	4,171	733	61,355
1960-1964	4	2	8	1	19	3	1,484	20,219	13,508	5,770	4,111	333	1,484	4,276	792	52,014
1965-1969	1	2	--	--	5	--	941	26,662	13,489	3,465	2,439	224	898	3,241	1,141	52,508
1970-1975	--	6.2[8]	--	--	2	--	146	17,428	13,410	1,224	2,619	151	576	3,183	2,149	40,893

[1] Species that have had an annual production greater than 1 million pounds.

[2] U.S. catch only until 1952

[3] Catches of walleys and blue pike combined through 1914

[4] Includes bullhead through 1951.

[5] Species normally less than 1 million pounds (goldfish, bullheads, burbot).

[6] Average for years of record

[7] Probably composed of 8 to 9 million pounds of blue pike and the remainder walleys.

[8] Esox species.

Table 6. Summary of Fish Species in Lake Erie Suffering Recent Population Decimations (From Hartman, 1972, 1973; Applegate and Van Meter, 1970; Smith, 1977C).

Species	Suspected Contributing Cause of Population Decimation						
	Related to Eutrophication						
	Destruction of Summer Oxygen Regimes	Siltation[1,2]	Over-Exploitation	Tributary Damming[2]	Marsh Drainage[2]	Temperature Fluctuations	Hybridization
Lake Trout	x	x	x				
Sturgeon	x		x	x	x		
Northern Pike				x	x		
Lake Herring	x		x			x	
Whitefish	x	x	x			x	
Sauger		x	x	x			x
Blue Pike	x		x			x	x

[1]Resulting from both erosion and accumulating organic material produced by nutrient enrichment.

[2]Of particular importance in regard to availability of or access to suitable spawning grounds.

historical summaries of the Lake Erie fishery are available (Applegate and Van Meter, 1970; Hartman, 1972 and 1973) while the raw data on volume and value of fishery products are summarized annually since 1970 by the National Marine Fisheries Service, and previously by the Bureau of Fisheries of the Fish and Wildlife Service. A recent GAO report (Government Accounting Office, 1977) contains a table (Table 3, App V) which summarizes the Lake Erie catch statistics by species since 1879. The Appendix contains a list of the common and scientific names of fish species mentioned in the report.

Prior to a recognized phosphate-induced eutrophication problem in Lake Erie, severe changes were evident in the fish community. Populations of several valuable species suffered decimations prior to 1930. Over-exploitation of the fish stock and elimination of suitable stream spawning grounds, or access to them, are usually blamed for the initial decline of these populations. Environmental degradation resulting from phosphate-induced eutrophication has further contributed to the biological extinction of some populations and has prevented the comeback of others in Lake Erie.

Lake trout populations in Lake Erie were decimated by overfishing in the late 19th century. The probably small population of slow-growing, late-maturing, long-lived fish could be expected to be eliminated by the fishing pressure of the turn of the century. The complete elimination of the species, however, was after 1937 (Smith, 1972c) and probably due to the destruction of suitable habitat for growth and reproduction. Nutrient enrichment, contributing to the destruction of summer oxygen regimes and siltation of spawning grounds, is the probable explanation for the biological extinction of lake trout in Lake Erie. Lake trout are currently being stocked in Lake Erie in small numbers (Zarbock, 1977), but there is no evidence that the population is sustaining.

Sturgeon populations declined rapidly between 1895 and 1919, and steadily after that time. The sturgeon is considered a rarity in the lake today. When the fish was abundant, little of the flesh was marketed; the roe was important for caviar, the bladder used in the manufacture of isinglass, and the carcass for oil. Large numbers were simply destroyed by fishermen because the large, powerful fish frequently caused heavy damage to fishing gear. Since sturgeon mature very slowly (some spawn for the first time at age 20), over-exploitation is the probable cause of their initial decline. Other factors, however, have also contributed: destruction of the bottom habitat by industrial, agricultural and domestic pollutants, and the damming of tributaries and drainage of marshes, with the consequent obstruction of spawning runs. Smith (1968) noticed that even with fishing pressure aleviated by partial protection, Lake Erie's population has continued to decline. He blames the inhibited recovery on the continued pollution of shallow spawning areas and eutrophication (S.H. Smith, personal communication).

The third abundant species eliminated in Lake Erie was the lake herring or cisco, whose numbers decreased markedly in 1925 and never recovered, except for a brief comeback in 1945-46. It has been commercially extinct since 1957. Landings of 10 to 50 million pounds were recorded in the 10 years prior to the collapse. Overfishing and possibly environmental change may have interacted to produce the sharp decline in lake herring stocks in the 1920s. The subsequent reduction in the mid-1960s was more likely a result of environmental stress (e.g. the high oxygen demand at the mud-water interface where the eggs incubate), as evidenced by the slow growth of the fish shortly before this time (Smith, 1968 and unpublished manuscript). Beeton and Edmondson (1972) proposed that overfishing was not the major cause for the population declines of lake herring. Rather they stated that pollution or eutrophication, or both, was probably responsible for the destruction of suitable habitat in parts of the lake, tending to concentrate populations in smaller areas where exploitation could continue.

Northern pike, the fifth most abundant species landed in Lake Erie in 1908, is also now commercially extinct. Destruction of natural spawning grounds, due to the construction of dams in the early 1900's, and drainage of marshes for farmland is the most likely cause for the collapse. Enrichment of shallow spawning areas can be expected to keep the population low (S.H. Smith, personal communication).

Whitefish and blue pike, too, have been impacted by eutrophication. Annual whitefish catches fluctuated between 1 and 7 million pounds from 1915 to 1954, and have gradually declined to 1-10 thousand in the past 10 years. Intensive exploitation is a significant factor in the decline, but environmental conditions, especially temperature and oxygen levels were undoubtedly important in recent decades. The whitefish require cold, well-oxygenated bottom waters in summer, and silt-free lake spawning areas for successful reproduction. Like the lake trout, it is at the southern end of its zoogeographical range, and even natural temperature cycling could have had a severe impact on the success of some year classes. Nutrient enrichment, especially in the western and central basins of Lake Erie, has eliminated suitable whitefish habitat through the depletion of summer oxygen regimes, as well as the sedimentation of decaying algae on spawning sites. Only remnants of the population remain in the deep eastern basin, which is least affected by eutrophication and intensive fishing.

Blue pike landings also declined precipitously in the late 1950s. That population, while extremely important in the Lake Erie fishery, naturally experienced wide fluctuations in harvest, ranging from 2 to 26 million pounds annually in the years from 1915 to 1957. Since that date, blue pike have declined to less than 1000 pounds in the last decade, and are apparently now biologically extinct (Hartman, 1973; Nepszy, 1977). Although subjected to intensive exploitation, the rapid decimation of the population implies an additional severe stress. The first recordings of oxygen depletion in the central basin coincided with the blue-pike decline of the late 1950s. Blue-pike were most

plentiful in this basin which was also their major spawning area (Smith, 1972c). Nepszy (1977) suggested that summer oxygen depletion in the central basin may have forced the blue pike to concentrate in the more suitable waters of the eastern basin, where they may have been more vulnerable to exploitation. He believed that exploitation and habitat destruction at critical times were the most serious stresses on the blue pike population, while rainbow smelt may have been another factor. The final blow may have resulted from hydridization with the walleye (Regier et al., 1969).

With the elimination of large populations of valuable fish, the fishery began in 1960 to depend more heavily on yellow perch, walleye, and smelt. Since that time, walleye and yellow perch populations have been declining. Landings of walleye peaked at 16 million pounds in the late 1950s, but fell precipitously in the 1960s, averaging 1.4 million pounds annually between 1960 and 1964, and 0.9 million pounds annually between 1965 and 1969. Since 1970, landings have not exceeded 252,000 pounds and in 1975 walleye production had been reduced to 2 percent of that in the peak year of 1956 (Schneider and Leach, 1977). Increased Canadian fishing pressure and catch efficiency with the use of sonar, nylon gill nets and ship-to-shore radio are blamed for the rapid decline in walleye catches. Populations in the western and central basins have also been impacted by the depletion of summer oxygen levels in the hypolimnion, siltation of spawning areas and losses in valuable benthic food organisms (e.g. Hexagenia) (Schneider and Leach, 1977). During the years of peak walleye production the bulk of the walleye catch came from the western and central basins while the contribution from the eastern basin was a small but consistent portion of the total catch. After the population decline and fishing bans in the western and central basins, the small, but persistent production from the eastern basin came to make up the bulk of the walleye catch. According to Schneider and Leach (1977) exploitation and nutrient loadings were the most important factors in the decline of walleye populations in the western basin. Smelt, introduced into an inland lake in Michigan in 1912, may also have caused

additional stress on walleye populations (Reiger et al., 1969), but alewife and sea lamprey populations were too sparse to impact walleye (Schneider and Leach, 1977). Strong walleye year classes in 1970 and 1972 indicate that the environment still allows reproduction to occur. According to recent evidence, walleye stocks are apparently recovering; commercial fishing bans in Ontario, Ohio and Michigan due to mercury contamination may be partially responsible (Nepszy, 1977).

Yellow perch has regularly been an important species in the Lake Erie fishery. After decimations of the whitefish, lake herring, and blue pike populations, it became far more important. Since 1955, yellow perch has dominated the annual landings; their numbers reached an all-time high in 1969 with 33 million pounds landed but declined to 10 million in 1975. The last highly succesful catch was in 1965, and in 1976, Ohio reported the lowest catch in the past 15 years (Zarbock, 1977).

Smelt was the most abundant fish in the commercial catch from Lake Erie in 1975. In the 15 years before that time it ranked second, behind yellow perch. Smelt is harvested primarily in Canadian waters; there is evidence that this area of the lake is preferred habitat. Smelt has been used for farm-animal food, but its value for human consumption is increasing. S.H. Smith (personal communication) noted that developments in the U.S. industry, eg. trawling and advanced processing plants, would do much for the U.S. smelt fishery.

White bass is expected to become a more important resource as the abundance of high-value species declines. Sheepshead (freshwater drum) populations are currently underexpolited because market acceptance of this species has been limited. Sheepshead are estimated to exceed all other fish species in biomass, but there may be greater numbers of shiners and gizzard shad. Channel catfish has supported a relatively stable fishery. A good market for live catfish exists but there is some evidence that populations are being fished too heavily. Low market

demand has impaired exploitation of populations of coarse fish such as carp, goldfish, suckers, and bullheads, yet the carp may rank second in biomass in the western section of the lake where it thrives in enriched waters.

Commercial Fishery - The Industry

Data contained in the Fishery Statistics of the United States (National Marine Fisheries Service) and a review by Applegate and Van Meter (1970) were used to analyze the changes in the Lake Erie fishery in the past 25 years. Several striking trends in the fishery are obvious.

The combined Canadian and United States production from Lake Erie has changed little over the past quarter century. Fluctuations in the tonnage landed are considered normal. On the other hand, the amount of fish taken from Canadian waters has increased remarkably, while the current United States catch has declined to less than one-third of its catch in the early 1950's. American fisherman caught 55% (25.5 million pounds) of the fish taken from Lake Erie during the period from 1950-1955; the same figure for 1970-1975 is 21.2% (8.82 million pounds). There are several explanations for the decline in the American fishery. The rapid expansion of the Canadian fishery around 1950 is one explanation for the increase in percentage taken by Canadian fishermen. Although not easily documented, declining catches in U.S. waters may be an indication that the preferred habitat of the most important fish is in Canadian waters. There is some evidence that suggests this might be true for smelt. Less restrictive regulations on harvest methods in Ontario and lower labor costs have fostered a sizeable smelt fishery there (W.L. Hartman, personal communication).

While the cause of the decline in the U.S. fishery is unclear, it is obvious that the fishery is not nearly as important as it was 25 years ago. The dollar value of the catch decreased along with the tonnage and the marketability of the fish species in the late 1960s but

since then the value has doubled ($2.7 million in 1976), even though the poundage has remained about the same. In Lake Erie, low value species, usually discarded from the catch, comprised 37% of the catch-by-weight in haul seines and 26% of that in trap nets in 1969. It has been estimated that an additional 2.8 million pounds could have been harvested from the lake if a suitable market for these fish existed (Zarbock, 1977). Yellow perch, although important, was long considered of secondary value when compared to species such as walleye, whitefish, and blue pike. With the elimination of the more valuable species, producers have come to rely heavily on yellow perch to remain in business. Smelt are largely taken only in Canadian waters; the increase in their value for human consumption has done little for American fishermen. Sheepshead (freshwater drum) have traditionally been considered a noxious fish and are usually sought by fishermen only when other, more valuable species are unavailable. Because of difficulties in marketing the fish for human consumption, sheepshead is one of the most underexploited major populations in the lake. The development of new markets for this abundant fish would help fishermen economically and perhaps benefit the more desirable walleye, yellow perch, and white bass; it is felt that sheepshead compete for space with these species. Fishing effort for white bass, once thought an incidental species, is expected to increase. The market for channel catfish has been stable for the past 20 years and the price paid for the landed fish is stable, as well. Carp, goldfish, suckers and bullhead are in low demand and the market is subject to unstable prices (Applegate and Van Meter, 1970). The less desirable, low-priced species are marketed for pet food processing but can no longer be used as food for captive fur bearing animals (e.g. mink) because of PCB contamination.

The impact of the declining Lake Erie fishery in the U.S. is reflected by the number of people and vessels employed in catching fish. From 1950 to 1962, the industry employed between 1,132 and 855 (mean, 963) regular, part-time, and casual fishermen. The total gross tonnage of fishing vessels ranged from 1,620 to 1,214 (mean, 1388) for the same

period. Since that time, the average number of fishermen decreased to 392, and average total vessel gross tonnage decreased to 488 for the years 1970-1972. In Ohio and Pennsylvania, the number of plants engaged in the processing and wholesale manufacture of fish products decreased by the early 1970's to less than half (mean, 33) of the number operating in the 1950's (mean, 73), but the number of people employed in these activities remained fairly constant.

Applegate and Van Meter (1970) warn that unless markets for the medium and low-value fish can be found, U.S. fishermen on Lake Erie are destined for bankruptcy. Yellow perch, although abundant, will not be able to keep the industry solvent. The loss in the fishery resource attributable to eutrophication alone is impossible to quantify in dollar value. It cannot be denied, however, that eutrophication of Lake Erie has had a very significant effect on the U.S. fishing industry and is at least partially responsible for its decline. While the significance of eutrophication in past fishery depletions remains to be quantified, Smith (1972a) recognized the importance of favorable water quality to the maintenance of sustaining fish stocks in the future. Fishery restoration programs in the Great Lakes will futile unless adequate measures are taken to create the proper water quality conditions necessary for the maintenance of fish stocks. Eutrophication is one of several threats to the success of fishery restoration programs in the Great Lakes.

Effects on Sport Fishing and Other Forms of Recreation

Sport fishing trends are far more difficult to follow because of the paucity of uniform regional data. Because of the recent growth of sport fishing, more interest is being expressed in a standarized system for sport fishing data within the Great Lakes region. Of the data available, as reported to the Great Lakes Fishery Commission by state and provincial agencies, yellow perch comprised 60% of the total sport catch in the Michigan waters of Lake Erie in 1975, and 47% of the sport

catch in Canadian waters of the lake in 1976 (Zarbock, 1977). Walleye and smallmouth bass made up 13% and 10% of the sport catch, respectively, in the Canadian waters of Lake Erie in 1976, and were the most sought after fish. White bass, freshwater drum and sunfish were also caught.

There is no available information on the impact that nutrient enrichment has had on the sport fishery. Traditionally, however, nutrient enrichment has been a standard management technique in parts of Europe, Asia and Africa where the production of commercial fish in small lakes and ponds is important. The addition of limited amounts of fertilizer is supposed to increase the productivity of the ecosystem and ultimately result in a larger carrying capacity of harvestable fish. In the United States, fertilization of lakes and ponds is usually employed for the production of sport fish and primarily in the southeast where the soils and lakes are infertile. Fertilization of lakes and ponds outside of this area is not a widely accepted practice. The problems associated with intentional fertilization, i.e. nuisance algal blooms and oxygen depletion resulting from algal respiration and/or decomposition, are the same as those associated with cultural eutrophication (Bennett, 1971). In many areas where artificial fertilization has been attempted salmonid (trout) and coregonine (ciscos and whitefish) fishes, which require cold, well-oxygenated water, have been eliminated. Many of the changes resulting from artificial fertilization are harmful to the most valuable sport fish (Tanner, 1960). Bennett (1971) recommends the use of artificial fertilization only in waters draining infertile soils.

Neil (1975) made some attempt to measure the effect of Cladophora on the recreational value of the lower Great Lakes. Cladophora is a filamentous alga whose growth is dependent upon nutrient enrichment. Cladophora affects beach and water uses when it is loosened from its attachments on rocks by wind and waves and washed up on the beach. Small dipterous flies and odors are associated with the drying and decaying algae. Many beaches along the lower Great Lakes are affected,

and if the beaches are not cleaned, many are not used. A survey of shore-line state parks was conducted, and estimates for clean-up operations related to Cladophora problems ranged up to $14,400 per park per year. Additional costs associated with swimming pool construction and shoreline improvements necessitated by eutrophication problems cannot be determined.

Data are not readily available on the number of sports fishermen, boaters, and swimmers affected by eutrophication, or their economic importance. Stroud (1977) estimated that sport fishing in the Great Lakes alone put approximately $149 million into the economy in 1975. An obvious trend has been the increase in the number of people enjoying water-based recreation requiring clean, unpolluted water.

Concurrently, there has been an increase in the amount of money people are willing to spend to protect or improve local water quality. One indication is the Clean Lakes Program which was established by PL 92-500 and authorized $300 million in matching funds to community projects for lake restoration and pollution control. Close to 56 projects have been funded so far, at a total cost to local communities of about $15 million. In the early 1960's, voters in Seattle, Washington, elected to use an estimated $120 million in public funds to divert sewage away from Lake Washington in an effort to improve the trophic status of the lake (Edmondson, 1973). The diversion of industrial and municipal wastes from Lake Sammamish, Washington cost $3 million (Welch, 1977). Many thousands of dollars are spent annually by private citzens or local governments to correct eutrophication problems. The amount that citzens are willing to spend on remedial projects such as weed harvest and copper sulfate treatments is some indication of how much a clean lake is worth. It was estimated that for Lake Sebasticook, a lake in Maine plagued with nuisance algal problems, $40,000 (1965) dollars would be required for annual copper sulfate treatment to control the algae (Anon, 1965). Mackenthun (1973) estimated that annual losses from nutrient-induced aquatic plant growths or their control have amounted to

$14 million for 17 western states. It is unfortunate that similar figures are not available on a nation-wide basis. These examples are good indicators of the importance of clean water to lake-shore communities.

VI. The Extent of the Eutrophication Problem

The preceding sections have delineated the impact of eutrophication. For complete evaluation, the magnitude of the eutrophication problem should be assessed. Three major indices can be used for this evaluation: the National Eutrophication Survey, the annual National Water Quality Inventory, and the National Commission on Water Quality. Each of these reports involves one (or several) scheme(s) for classifying the trophic condition of lakes. There is a wide diversity in the schemes available. While no attempt has been made in this report to evaluate the different schemes, it should be recognized that lack of a uniform classification system interferes to some extent with an attempt to estimate the scope of the eutrophication problem. Nevertheless, the information to follow is the best available means of evaluating the extent of the problem. Another caution is necessary. The following evaluation makes no attempt to differentiate between cultural and natural eutrophication. While the vast majority of eutrophication problems are related to human activities, high natural phosphate levels and low anthropogenic loadings make natural eutrophication a problem in some areas. For example, Florida and Vermont reported in the 1975 National Water Quality Inventory (U.S. Environmental Protection Agency, 1975) that natural eutrophication would prevent them from meeting 1983 goals of fishable and swimmable waters. Both states also recognize the significance of anthropogenic sources of phosphorus.

The National Eutrophication Survey was initiated in 1972 by EPA to identify and characterize lakes receiving municipal sewage effluent; it was later expanded in scope to include lakes receiving non-point source contributions. The data are not necessarily representative of the

conditions of all lakes and reservoirs in the United States. Consequently, they have only limited value for quantitative analysis of the extent of eutrophication, but they can be used as a guide in determining the scope of the problem.

Potential eutrophication problems could be expected if median total phosphorus concentrations in a lake exceeded 0.025 mg/l or if annual phosphorus loadings exceeded a critical level as defined by Vollenweider (1968). Data from 298 lakes in 22 states east of the Mississippi were used for this assessment. Most (73%) of the lakes for which data were available exceeded the 0.025 mg/l total phosphorus level. The majority of these lakes (85%) were impacted by sewage effluents. Other nutrient sources, however, were equally important in most of the lakes. More than 74% of the 234 lakes for which loading analysis was completed received less than half of the total phosphorus load from municipal waste. Using Vollenweider's model, 83% of 156 lakes received total annual phosphorus loadings at rates characterized as eutrophic. Sewage enriched lakes had a greater incidence of eutrophication (82%) than lakes impacted from only non-point sources (30%) (U.S. Environmental Protection Agency, 1975).

Omernik (1977) mapped the nutrient concentrations found in 928 stream sites sampled as part of the National Eutrophication Survey. Only sites impacted solely by non-point sources were considered. The map shows a remarkable concentration of areas exceeding the "safe" phosphorus level for flowing waters (>.1 mg/l as total P) in central California and the Great Plains (North and South Dakota, Nebraska, Kansas, Oklahoma, Iowa, southern Minnesota, Illinois, Indiana and Ohio). In fact most (38) of the contiguous United States contained at least one area with stream phosphorus concentrations exceeding the safe limit, and all but one of the states approached (0.071-0.1 mg/l total P) the safe level. In an accompanying report, Omernik (1976) found significantly higher nutrient content in streams draining agricultural watersheds versus streams draining forested watersheds. Omernik's map shows how

widespread nutrient enrichment is in streams impacted only by non-point sources, but it gives no indication of water quality problems resulting from enrichment. If we were to exclude the areas receiving high natural phosphorus loadings (e.g. central Florida) and add the streams impacted by point sources, the areas experiencing high levels are likely to be even more extensive. Unfortunately, the data needed for this analysis have not been compiled.

The National Commission on Water Quality, established by PL 92-500 section 315, attempted to assess the range of water pollution problems throughout the United States and how point source controls would affect them. The country was divided into 13 regions on the basis of the principal geologic, soil, and vegetation types. A total of 41 sites were selected within the regions on the basis of available environmental data. Water quality problems existed at most of the locations selected for study, but most areas contained both polluted and non-polluted sections, and a number of sites with relatively minor water quality problems were analyzed. Some areas were affected by both point and non-point source pollutants.

Twenty-two (54%) of the 41 sites reported major problems with high nutrient levels (nitrate nitrogen $>$ 0.9 mg/l, total P$>$0.1 mg/l) and the symptoms of eutrophication. Ten additional sites reported sufficiently high nutrient levels, but algal blooms were limited by turbidity or flow velocity. The majority of sites with major nutrient-enrichment problems were impacted primarily by municipal wastewater discharge. High nutrient levels and eutrophication problems were considered significant in 5 of the 13 regions: Midwest, Great Lakes, Middle Atlantic/South/Gulf, Southern Plains, and Central Valley. Nutrients and eutrophication problems were of moderate significance at sites in the Northern Plains, Columbia Plateau, Hawaii, Great Basin and Southwest Desert, Colorado Plateau, Upper Ohio, New England and Puerto Rico (Figure 7). (National Commission on Water Quality, 1976).

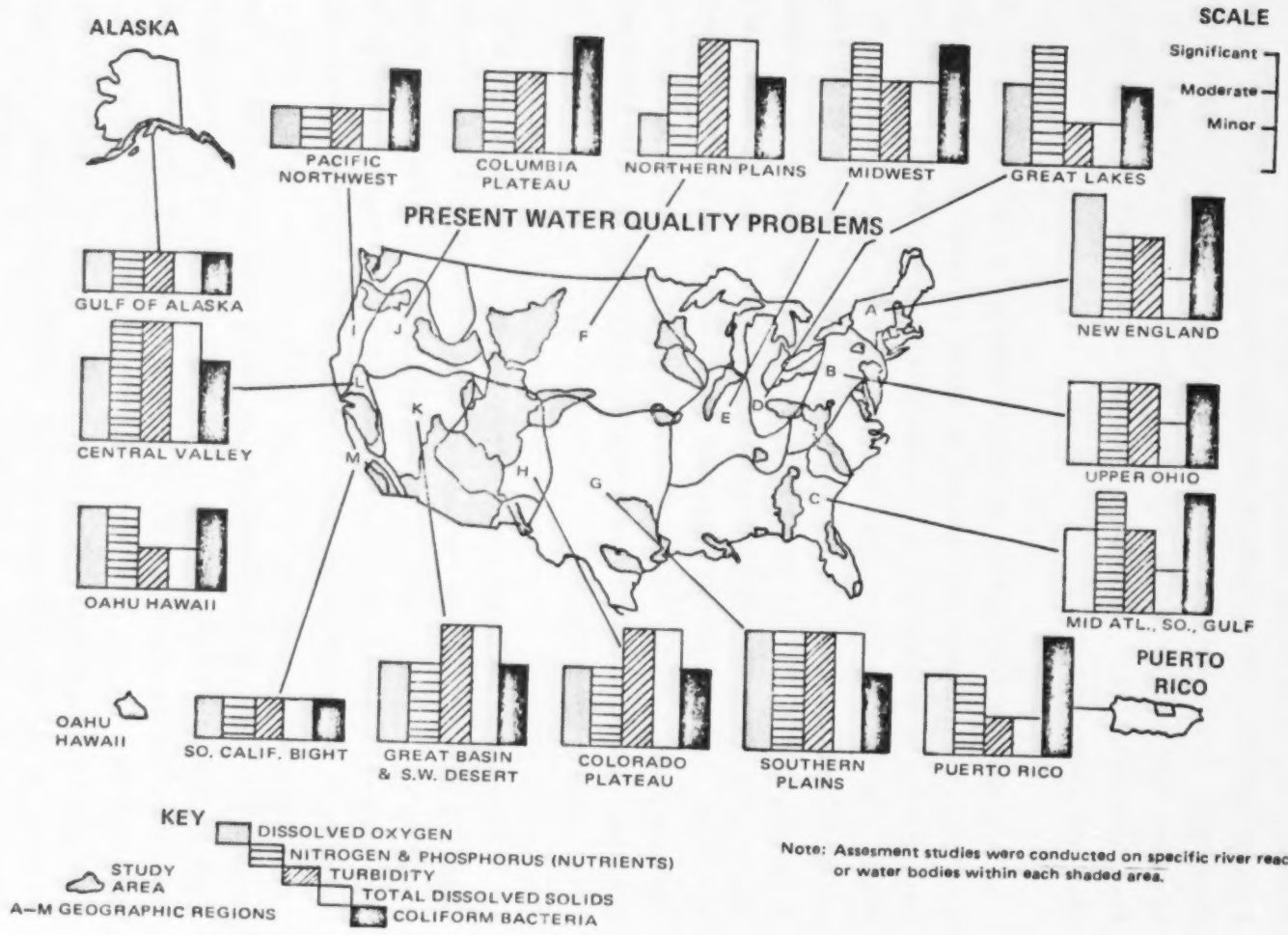

FIGURE 8. RELATIVE IMPORTANCE OF SELECTED WATER QUALITY PROBLEMS IN THE UNITED STATES. (NATIONAL COMMISSION ON WATER QUALITY, 1975).

The most recent National Water Quality Inventories (1975 and 1976) contain information which can be used to assess the extent of the eutrophication problem. The annual Inventory is mandated by section 305(b) of the Federal Water Quality Act Ammendments of 1972. It summarizes state reports which describe the water quality of all navigable waters within the state.

The 1975 Inventory (U.S. Environmental Protection Agency, 1975) reports that high (P concentration not specified) phosphorus and nitrogen loadings, indicative of a potential eutrophication problem, were mentioned as a problem in 43 of the 52 state reports. The 43 states reporting high nutrient levels were distributed across the country; relatively fewer southern and southwestern states reported problems in comparison with the Great Lakes, central, western, middle Atlantic, and northwestern states. Ten states reported lower nutrient levels and decreasing eutrophication potential; 28 states reported no change in nutrient levels; five states, however, reported worsening eutrophication potential. Despite nutrient and other water quality problems, most (23) of the 32 states attempting an overall evaluation reported good water quality overall.

The 1976 state reports indicate that high nutrient concentrations are still a problem in many states. Maryland, Vermont, and Florida cite high levels of phosphorus and nitrogen as the most serious water quality problem (U.S. Environmental Protection Agency, 1976b).

Other states have attempted to classify the trophic status of their lakes, as mandated by Section 314(a) of PL 92-500. State reports for Wisconsin and Michigan contained readily usuable information for this assessment.

In Wisconsin, Uttomark and Wall (1975) classified 98% of the lakes equal to or greater than 100 acres in size. They developed a "Lake

Classification Index" based on several observable parameters indicating eutrophication, including oxygen level of the hypolimnion, transparency, history of fish kills and recreational use impairment by excessive aquatic plant growth. Each lake was assigned "penalty points" which increased with the degree to which undersirable characteristics were exhibited. According to this index, 30% of the lakes analyzed were oligotrophic, 50% were mesotrophic and 20% were eutrophic. In comparison with the National Eutrophication Survey, which surveyed 42 of the same lakes, the LCI is more optimistic, as shown in Table 7 (Wisconsin Department of Natural Resources, 1976).

Table 7. Trophic Status of a Set of Wisconsin Lakes, as determined by Uttormark and Wall (1975) and the National Eutrophication Survey (U.S. Environmental Protection Agency, 1975)

Lake Type	Uttormark & Wall # of Lakes	%	NES # of Lakes	%
Oligotrophic	5	12	2	5
Mesotrophic	16	38	5	12
Eutrophic	21	50	35	83
	42		42	

Michigan (Michigan Department of Natural Resources, 1976) surveyed 15% of the lakes more than 50 acres in size, representing nearly half of the state's total combined lake surface area. Trophic status for each of the 286 lakes was determined using two methods, depending on the degree of study. If a lake met three of the following criteria, it was considered eutrophic:

1. Clinograde oxygen curve during stratification.
2. Dissolved oxygen $\leq$ 0.5 mg/l 1 meter off the bottom during stratification.
3. Secchi disc transparency $<$ 2m.
4. Chlorophyll a $>$10 ug/l.

5. Organic nitrogen >0.8 mg/l.
6. Total inorganic nitrogen >0.3 mg/l (NH_3 + NO_3)
7. Total phosphorus >0.10 mg/l.

Alternatively, citzens participated in a Self-help Water Quality Monitoring Program for some lakes and considered a lake eutrophic if the secchi disk transparency was less than 2 meters, or the chlorophyll a concentration greater than 10 ug/l. National Eutrophication Survey criteria were used in NES lakes. Of those surveyed, lakes in more heavily populated areas were more likely to be found eutrophic: 24.6% were considered eutrophic in the upper Peninsula, 27.4% in the northern portion of the lower Peninsula and 56.4% in the southern portion of the lower Penisula. Based on these data, 39.5% (776) of the large (>750 acres) lakes in Michigan are eutrophic, although small sample size and purposeful selection of problem lakes may bias this extrapolation.

APPENDIX

The Appendix includes the scientific names and some colloquial names given to the fishes mentioned in this report. The names are listed alphabetically under the common name adopted by the American Fisheries Society (AFS) (1970). The asterisk denotes European fishes not included in the AFS publication.

Common Name	Scientific Name
Alewife	*Alosa pseudoharengus*
Atlantic salmon	*Salmon salar*
*Blaufelchen	*Coregonus wartmanni*
Blue pike	*Stizostedion vitreum glaucum*
Bullhead, including:	
Brown bullhead	*Ictalurus nebulosus*
Yellow bullhead	*Ictalurus natalis*
Black bullhead	*Ictalurus melas*
Burbot	*Lota lota*
Carp	*Cyprinus carpio*
Channel catfish	*Ictalurus punctatus*
Cisco (lake herring)	*Coregonus artedii*
Deepwater cisco	*Coregonus johannae*
*Eurasian perch	*Perca fluviatilus*
Fourhorn sculpin (deepwater sculpin)	*Myoxocephalus quadricornus*
Freshwater drum (sheepshead)	*Aplodinotus grunniens*
Gizzard shad	*Dorosma cepedianum*
Goldfish	*Carassius auratus*
Lake sturgeon	*Acipenser fulvescens*
Lake trout	*Salvelinus namaycush*
Lake whitefish	*Coregonus clupeaformis*
Mummichog	*Fundulus heteroclitus*
Northern pike	*Esox lucius*
Rainbow smelt (smelt)	*Osmerus mordax*
Sauger	*Stizostedion canadense*

APPENDIX (cont.)

Common Name	Scientific Name
Sea lamprey	*Petromyzon marinus*
Sheepshead minnow	*Cyprinodon variegatus*
Shiners, including:	*Notropus* spp. and *Notemigonus* spp.
Emerald shiner	*Notropus atherinoides*
Golden shiner	*Notemigonus crysoleucas*
Smallmouth bass	*Micropterus dolomieui*
Sockeye salmon (kokanee)	*Oncorhynchus nerka*
Suckers, including:	*Catostomidae*
Quillback	*Carpoides cyprinus*
Norther redhorse	*Moxostoma marcolepidotus*
Silver redhorse	*Moxostoma anisurum*
Golden redhorse	*Moxostoma erythrurum*
White redhorse	*Moxostoma commersoni*
Sunfish	*Lepomis* spp.
Walleye (yellow pike, pickerel)	*Stizostedion vitreum vitreum*
White bass	*Morone chrysops*
Yellow perch	*Perca flavescens*

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